A spotted landscape: Threats to leopard, *Panthera pardus pardus*, & their prey within the Boland Mountain Complex, Western Cape

by

Brittany Claudia Schultz

Thesis presented in partial fulfilment of the requirements for the Degree of

Master of Science

at

Stellenbosch University

Department of Conservation Ecology and Entomology,

Faculty of AgriSciences

Supervisor: Dr. Alison J. Leslie
Scientific Advisor: Anita Wilkinson

March, 2020

Declaration

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

Signed: Brittany C. Schultz
Date: March, 2020.

Abstract

The collapse of prey-bases threatens many predators globally and may contribute to some predators' localised extinctions. A similar cascade is a potential threat to leopard *Panthera pardus* and their medium-sized mammalian prey populations in the Fynbos biome. Medium-sized mammals have reportedly been negatively impacted by a number of anthropogenic threats in agricultural land-covers that act as buffers between human development and natural fynbos habitats. One of these threats and a driver of many, is the loss of habitat from human-caused land-cover changes. The Boland Mountain Complex (BMC) is one of the eight patches of protected mountainous areas, proclaimed as a United Nations Environmental, Educational, Scientific and Cultural Organization's (UNESCO) World Heritage Site, in the Western Cape Province, South Africa. The BMC forms a key part of the leopard's range within the Fynbos biome and has a relatively high diversity of medium-sized mammalian species, which utilise both the core protected areas and the surrounding agricultural buffer zones. Multiple adjacent human settlements are development hotspots and have increasing human population sizes. The ecology of many of these mammal species has not been well studied, particularly in the Fynbos biome. It is therefore essential that a baseline study be conducted to determine where future research inputs should be focused to mitigate potential threats in the BMC area.

This study aimed to determine the extent that medium-sized mammals are threatened by human development in agricultural buffer zones in the BMC. Firstly, it was determined whether mammalian habitat was at risk of loss to land-cover change and shifts in fire regimes. Secondly, the study aimed to determine if there were apparent changes in any perceived, relative mammalian abundances over time, on agricultural buffer properties in the BMC. Geographic Information Systems (GIS) technology was used to analyse historic fire record data, obtained from the South African National Biodiversity Institute (SANBI) for the years 1957 to 2017. Further, two South African national land-cover datasets were obtained from the Department of Environmental Affairs (DEA), to analyse the land-cover changes for 1990 and 2013. Additionally, Local Ecological Knowledge (LEK) data, obtained through structured interviews with labourers and management stakeholders on agricultural properties, were gathered and analysed.

Land-cover changes displayed an overall increase in the "natural" vegetation cover class of 107.6km^2 mostly due to regrowth where plantations were felled, therefore remaining positively consistent over the 23-year period. The number of fires per year increased by five-times the average number that burnt per year from 1957 to 2017. Further, a three-fold increase in total burnt area was detected from 1972 to 2017, when compared to the 1957 to 1972 records. Of the eight species that showed significantly lower perceived relative-abundances in parts of the BMC, hares *Lepus spp.* and grey rhebok *Pelea capreolus* generated the greatest concern for their populations' survival. Detected threats that may be driving population changes in buffer zones, include: feral dogs, illegal hunting, edge effects and isolation of habitats due to land-cover change, roads and fencing. Differences in mammal compositions and frequencies of species' sightings, fire regimes and land-cover changes were seen between the defined mountain zones (MZ) for this area. This study thus provides considerations of mammal distributions and threats to the various species for future spatial and management planning.

Opsomming

Die ineenstorting van beskikbare prooi bedreig meeste roofdiere wêreld wyd en kan bydrae tot gelokaliseerde uitsterwe van hierdie roofdiere. In Fynbos biome kan ‘n soortgelyke effek ‘n potensiële bedreiging wees vir middelmatige grootte soogdiere en prooi populasies van die luiperd panthera pardus pardus, Middelmatige grootte soogdiere word volgens gerugte nadelig beinvloed deur ‘n verskeidenheid menslike bedreigings, onder andere landbou grond bedekkings, wat ‘n buffer sone veroorsaak tussen menslike ontwikkleling en die omliggende natuurlike Fynbos areas. Die verlies aan habitat as gevolg van menslike grond bedekking veranderinge is een van die hoof bedreigings en die katalisator van vele ander bedreigings. Die Boland Bergreeks Kompleks (BMK) is geleë in die Wes-Kaapse provinsie, Suid-Afrika, en is een van agt beskermde bergatige areas wat verklaar is onder die “United Nations Environmental, Educational, Scientific and Cultural Organization’s (UNESCO)” Wêreld Erfenis Gebiede. Die BMK vorm ‘n deel van die luiperd se kern area binne die Fynbos biome en het ‘n redelike hoë diversiteit van middelmatige grootte soogdier spesies wat hoofsaaklik gebruik maak van hierdie kern areas, die beskermde areas, asook die omliggende landbou buffer sones. Die omliggende area ervaar tans ‘n toename in ontwikkeling asook populasie groei. Die ekologie van baie van hierdie soogdier spesies was nog nie van tevore deeglik bestudeer nie, veral nie die in die Fynbos biome nie. Derhalwe is dit noodsaaklik dat ‘n fundamentele studie onderneem word om te bepaal wat die fokus van toekomstige navorsings projekte moet wees ten einde die impak van toekomstige potensiële bedreigings in die BMK te ondervang.

Die doel van hierdie studie was om te bepaal tot watter mate middelmatige grootte soogdiere bedreig word as gevolg van menslike ontwikkeling in landbou buffer sones, binne die BMK. Ten eerste, het die studie hom ten doel gestel om te bepaal wat die impak van veranderinge in grond bedekking asook veranderinge in brand patrone op soogdier habitate sal hê. Ten tweede was die doel van die studie om te bepaal of daar enige opmerklike veranderinge in die afname van die soogdiere, oor tyd op landbou buffer sones in die BMK was. Geografiese Inligting Stelsels (GIS) het die tegnologiese basis daar gestel, waarop die historiese data wat ontvang is vanaf die Suid Afrikaanse Instituut vir Biodiversiteit (SANBI) rakende brande vir die periode tussen 1975 tot 2017, ontleed is. Die Department van Omgewingswese (DEA) het twee addisionele Suid-Afrikaanse grond bedekkings data stelle beskikbaar gestel wat gebruik was om die grond bedekkings veranderinge te analiseer vir die periode tussen 1990 en 2013. Data (bekend as “Local Ecological Knowledge” (LEK)) wat saamgestel is deur formele onderhoude te voer met werknemers en bestuurspanne van belanghebbendes in die landbou bedryf was ontvang en ook ontleed.

Grond bedekkings veranderinge het ‘n algemene toename getoon in natuurlike plantegroei in ‘n area van ongeveer 107.6km2. Hierdie toename kan hoofsaaklik toegeskryf word aan die feit dat plantegroei weer gevestig het, waar plantasies afgekap was en gevolglik het grond bedekkings veranderinge redelik konstant gebly in die laaste 23 jaar. Jaarlikse brande het egter toegeneem teen ‘n spoed wat vyf keer meer is as die gemiddelde jaarlikse brande vir die tydperk tussen 1957 tot 2017. Wanneer die totale brand area vir die periode 1972 – 2017 vergelyk word met die vir die tydperk 1957- 1972, is dit opmerklik dat die totale brand area drievoud verhoog het. Die grootse bron van kommer vir die oorlewing van ‘n spesie was vir hase Lepus spp. en die grys rhebok Pelea capreolus, aangesien hierdie spesie, uit die agt spesies wat bestudeer was, die een was wat ‘n duidelike afname getoon het in relatiewe verspreidings in gedeeltes van die BMC. Verandering in populasie in buffer sones word bedreig deur onder andere, wilde honde, onwettige jag, isolasie van habitat as gevolg van veranderinge in grond bedekking, paaie en omheining.

Daar is verskille opgemerk in soogdier samestellings en die gereeldheidsbasis waarop soogdiere waarneem word, asook brand patrone en grond bedekkings veranderinge tussen die gedefinieerde bergreekse (MZ) vir hierdie area. Derhalwe voorsien hierdie studie oorweging van soogdier verspreiding en die bedreigings van die verskillende spesies vir toekomstige ruimtelike- en bestuursbeplanning.

Dedication

This thesis is dedicated to

My mother – Diedré Glenda Griffiths Schultz
My grandmother – Ernestine Huskisson Griffiths (1924-2012)

“Here’s to strong women. May we know them. May we be them. May we raise them.”

Acknowledgements

I would like to say thank you to my supervisor Dr Alison Leslie for her inspiration, guidance and so much patience throughout my MSc. Thank you to the Conservation Ecology and Entomology department of Stellenbosch University.

My gratitude is especially for Anita Wilkinson who was able to assist me at all levels and gave me constant support and experience throughout. Thank you to the Cape Leopard Trust for all the support, financially and logistically. Without their research, connections to landowners and inspiring passion for solving the human-wildlife conflict issues that face leopards and the Boland Mountain Complex's ecology, this project would have no grounds for inception.

I appreciate all the interviewees who participated in the study and contributed so much personal time and knowledge.

Thank you to my field work team member, Wian Nieman, for all the translations, conversations, laughs, sneezes, and advice.

Thanks to Prof Martin Kidd and Dr James Pryke for the statistics assistance.

Thank you to my parents, Diedre and Robin Schultz for inspiring my perseverance, feeding me and trying to understand and assist with my scientific field as much as you knew how. Thanks to my brother, Cameron Schultz for all the technical support.

Thank you to all my friends for the stress-relief and support during all the ups and downs. A separate thanks goes to my fellow Conservation Ecology friends for all the check-ins, needed breaks and venting sessions, you all made this experience whole and resolved the loneliness of it. Thanks to Andrea Grobler for the Afrikaans translations.

Thank you to Michael Brits for the hours of GIS assistance, emotional assistance and tough love when needed.

Table of Contents

List of Tables

List of Figures

List of Appendices

Chapter 1
Introduction and literature review

1.1 Leopards as umbrella species

Leopards, *Panthera pardus,* occur on an extensive number of both natural and human-dominated habitats on the planet (Hayward et al., 2006; Jacobson et al., 2016). Leopards make good flagship species because of their charisma that allows them to be conservation symbols (Mooney et al., 1995; Simberloff, 1998; Sinclair, 2003; Wang & Macdonald, 2009). They fulfil the typical role of keystone species because of how one individual or populations at seemingly low abundances, generally impact local community ecologies on multiple levels (Mooney et al., 1995; Lovari et al., 2009), like in the Fynbos Biome. In much of their range, they are often the only remaining apex predator, due to extirpation/extinction of other species (Athreya et al., 2013). They fulfil many vital ecological roles by influencing the behaviour, long-term behavioural adaptations and the distribution of prey species, many smaller species and competitors (Kerley et al., 2003; Martin & Harris, 2013; Ripple et al., 2014; Schneider, 2001). Like other predators, leopards can potentially regulate their prey population sizes, often in small, human-confined habitats, which may cascade into other trophic levels (Estes et al., 2011; Power, 2002; Peel & Montgu, 1999).

Leopards are recognised as umbrella species which are defined as species that, if protected, multiple other species which share resources and fall within the same ecological web, are then also protected (Noss, 1990; Simberloff, 1998). Their vital role as an apex predator combined with their significance to many local stakeholders, further justifies their function as flagship and umbrella species in the BMC to assess their ecology and anthropogenic threats in medium-sized mammals (Pitman et al., 2017; Wang & Macdonald, 2009). The use of umbrella species can be limiting when looking at smaller-scale protected areas (PA) or specific habitat types (Caro, 2003), since both above-mentioned scenarios often do not encompass the majority of other local species' population ranges (Caro, 2003). Leopards widespread home ranges (that encompass many other species' habitat requirements within them) make them ideal umbrella species in the BMC (Mooney et al., 1995; Martins & Martins, 2006; Simberloff, 1998; Wang & Macdonald, 2009).

Leopard species ranges have decreased by between 63% and 75% globally, and between 28% and 59% in Southern Africa, due to habitat loss and fragmentation, urbanisation and prey loss (Jacobson et al., 2016; Ray et al., 2005; Stein et al., 2016). It is often due to habitat loss and prey depletion that these predators are forced into situations of human-wildlife conflict (Marker & Dickman, 2005; Ripple et al., 2014; Woodroffe & Ginsberg, 1998). Both habitat loss and a decrease in habitat quality lowers the carrying capacity for various species, thus increasing the risk of extinction (Griffin & Drake, 2008). More severe and extensive effects occur when habitats that carry unique key-functions, harbour rare niches and are irreplaceable, are lost in a system that has been confined by human development (Brooks et al., 2002; Musakwa & Wang, 2018). In particular, the loss of corridors severs connections between leopard populations, which may convert a habitat fragment into a “sink” from a “source” population, resulting in the loss of individuals and a loss in genetic diversity (Brooks et al., 2002; Martins & Martins, 2006; McRae et al., 2008). These threats to habitats result in feedbacks on one another, creating a synergy of impacts on many species in an ecosystem (Khan et al., 2018). Globally, 50% of leopards prey species are classified as declining in abundance, most predominantly due to habitat change in the form of agricultural expansion (Wolf & Ripple, 2016). In sub-Saharan Africa’s protected areas, leopard prey has decreased by 59% (Stein et al., 2016).

Although many studies have looked at leopards globally, research on leopards in South Africa has been described as disproportional, with a limited range of topics (Balme et al., 2014; Gavashelishvili & Lukarevskiy, 2008; Jacobson et al., 2016). This lack of research is applicable to local leopard populations in the Fynbos biome, but they have been better studied than the majority of other local mammal populations (Martins & Harris, 2013; Martins & Martins, 2006). South Africa’s leopards have been ranked as “vulnerable” on the International Union for the Conservation of Nature’s (IUCN) red list since 2016 (Stein et al., 2016). This status has been re-evaluated several times as they were classified as “near threatened” and “least concern” in the last 10 years (Stein et al., 2016).

In South Africa, leopard ranges have been reduced to six populations in smaller patches, some are isolated and others share habitat with neighbouring countries (Jacobson et al., 2016; Swanepoel et al., 2013). The BMC’s leopards form part of the population that is loosely described as the Cape mountain leopards that occur in the mountainous regions in the Western and Eastern Cape, South Africa (Rautenbach, 2010). They are described as morphological variants of African leopard and are substantially smaller in size, weighing an average of 31kg (Martins & Martins, 2006; Radloff, 2008; Stein et al., 2016), when compared to the other, larger African leopard populations that weigh roughly twice that mass (Hunter et al., 2013). Baily (1993) described the trend of smaller sized leopards hunting smaller prey species, throughout Africa. Some have questioned whether leopards in the Cape evolved this smaller body size due to the prey and resource constraints of their habitat, which supports few larger faunal species at seemingly low abundances (Hayward, et al., 2006; Martins & Martins, 2006; Radloff, 2008). This naturally occurring scarcity of prey in the Fynbos biome, leaves leopards more vulnerable to any prey losses that may occur. Most predators’ physical and behavioural adaptations are determined by prey availability (Norton et al., 1986; Radloff, 2008; Radloff et al., 2010; Rautenbach, 2010). Greater prey biomass correlates positively to a higher density of leopard and where a lower prey density occurs, leopard ranges are larger (Baily, 1993; Marker &

Dickman, 2005; Odden & Wegge, 2005). Some leopard populations, like those in Phinda Game Reserve, Kwa-Zulu Natal have also shown hunting preference to specific habitat types, which can determine their predation rates more than the density of prey available (Balme et al., 2007). Both habitat and prey-base are therefore fundamental in maintaining functional leopard populations.

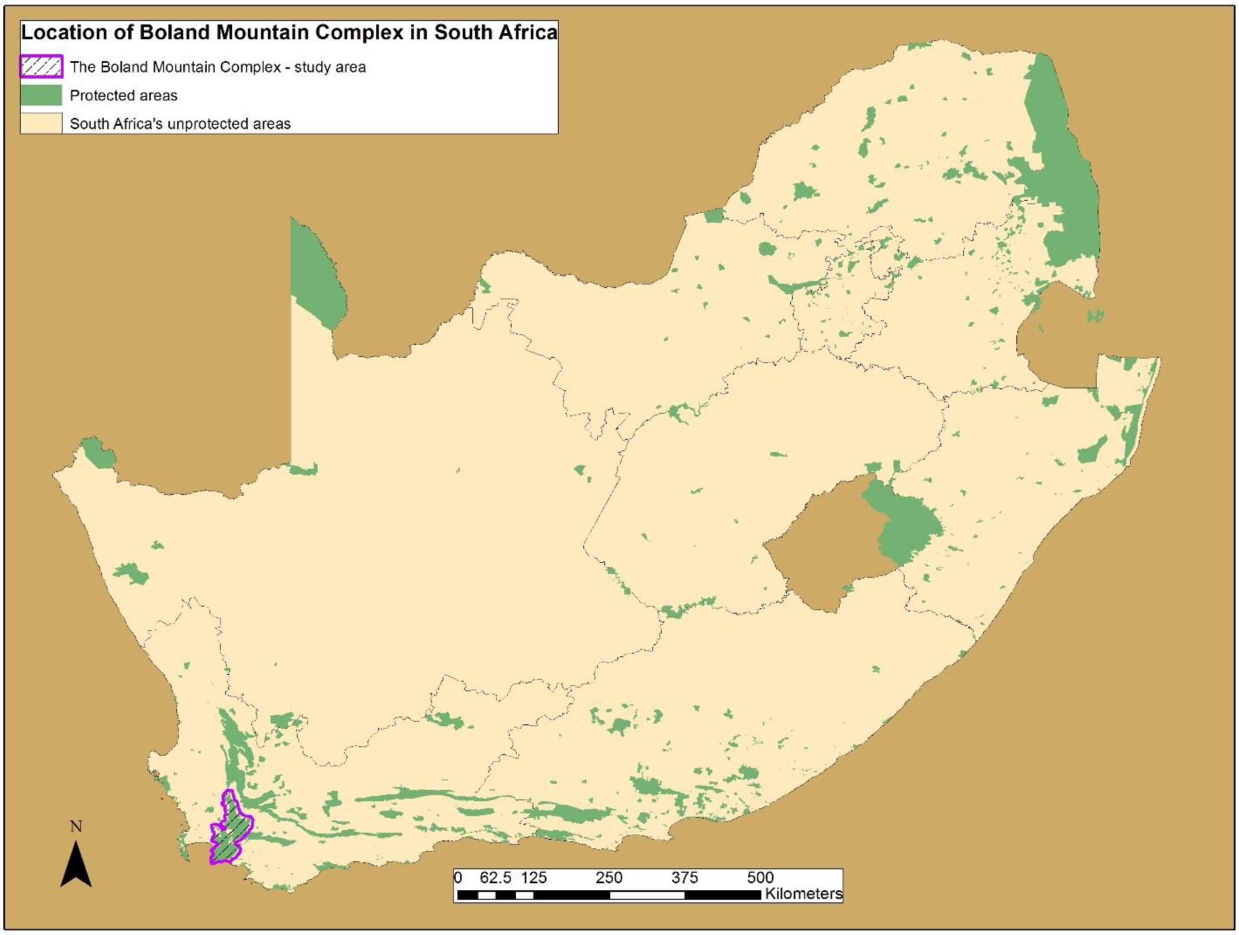
1.2 The Boland Mountain Complex

The majority of the Fynbos biome is found in the Western Cape province of South Africa which has an extremely high floral diversity and biodiversity in general (Rebelo, et al., 2006). The faunal diversity of the region is often over-looked, but consists of a plethora of invertebrate, bird, reptile and small to large mammal species (Birss, 2017; Rebelo et al., 2006). The province has a large human population that is continually growing and with it, both urban and agricultural expanses (Statistics South Africa, 2018). Within 100km east of the capital city of Cape Town is the BMC. This is one of eight protected United Nations Environmental, Educational, Scientific and Cultural Organization (UNESCO) Heritage Sites within the Fynbos Biome. The BMC overlays two adjacent UNESCO Biosphere Reserves, the Kogelberg Biosphere Reserve (KBR) and the Cape Winelands Biosphere Reserve (CWBR). It provides the most south-western remaining ranges of leopards in the Western Cape and contains 28 other medium-sized mammal species (Martins & Martins, 2006).

The study area was centred around the BMC to the extent of where it was assumed leopards (and therefore medium-sized mammals) could access the area (Figure. 1.1). It consisted of the three defined land-cover structures; core, buffer and transitional zones, of the UNESCO Biosphere Reserves. The core zones are entire PAs of natural habitat with limited human disturbances that make up the majority of the BMC. Buffer zones surround the core zones and are human disturbed, consisting of mostly agriculture and forestry land-covers in the study area (Boshoff et al., 2001). Buffer zones allow for usage of land by humans, and by various animals as corridors between core areas (Kintz, et al., 2006; Lovell & Johnston, 2009). The buffer zones integrate transitional and core zone functions, having negative and positive impacts on natural ecosystems (Makenzi, 2013; UNESCO, 1996). Transitional zones are concentrated human land-covers, containing infrastructure and/or intensive agriculture, although almost entirely anthropogenic habitats and not utilised by much wildlife, these zones' activities should aim to be sustainable (Makenzi, 2013). The transformed zones are sporadically located on the outer edges of buffer zones or as nodes surrounded by the buffer zones, within core regions.

The layout of the land-covers in the BMC form a matrix of human and natural land-uses, producing various impacts on faunal species (Mangall & Crowe, 2003). Threats to mammal biodiversity within this landscape include: persecution over human-wildlife conflict (Martins & Martins, 2006); edge effects (Elias, 2014); reduction in food sources; illegal hunting and trapping

(Nieman et al., 2019); physical and functional habitat loss (primarily from urbanisation and agricultural expansion) (Lombard et al., 1997); introduction of invasive predators (such as feral cats *Felis catus* and pigs *Sus scrofa*) (Botha, 1989; Morling, 2014); spread of disease and parasites (from humans, domestic animals and vehicles) (Roelke et al., 1993); unnatural, harmful fire regimes (Rebelo, 1992), and exposure to unnatural toxins and vehicle collisions (Serieys et al., 2019). Many of these listed threats start fundamentally from changes in land-cover sizes, human disturbances and contents from neighbouring developments (Martins &



Martins, 2006).

Figure 1.1: Location of the study area (purple outlined area) within South Africa and its official protected areas (green shaded areas within purple outline)

The landscape consists of multiple Cape Fold Mountain ranges with fertile valleys between them and the southern border is a 70km long coastline. The altitude ranges from 0m to 1994m (Johnson et al., 2006). The underlying geology of the area primarily consists of sandstone and granite intrusions (Johnson et al., 2006). The natural vegetation types (Figure. 1.2) include diverse groups of mostly fynbos and renosterveld which thrive in the Mediterranean climate, with great variation between seasons and landscape niches (van Wilgen, 1987). This generally consists of dry, hot summers, reaching above 40°C when fire is a frequent phenomenon. The winters are cold and wet, dropping below 0°C in the higher mountains, where snow falls yearly (Boucher & Moll, 1980; van Wilgen, 1987). The BMC contains many natural waterbodies including wetlands, rivers and the cold Atlantic Ocean in the south, which has strong effects on the localized climate (Conradie, 2012).

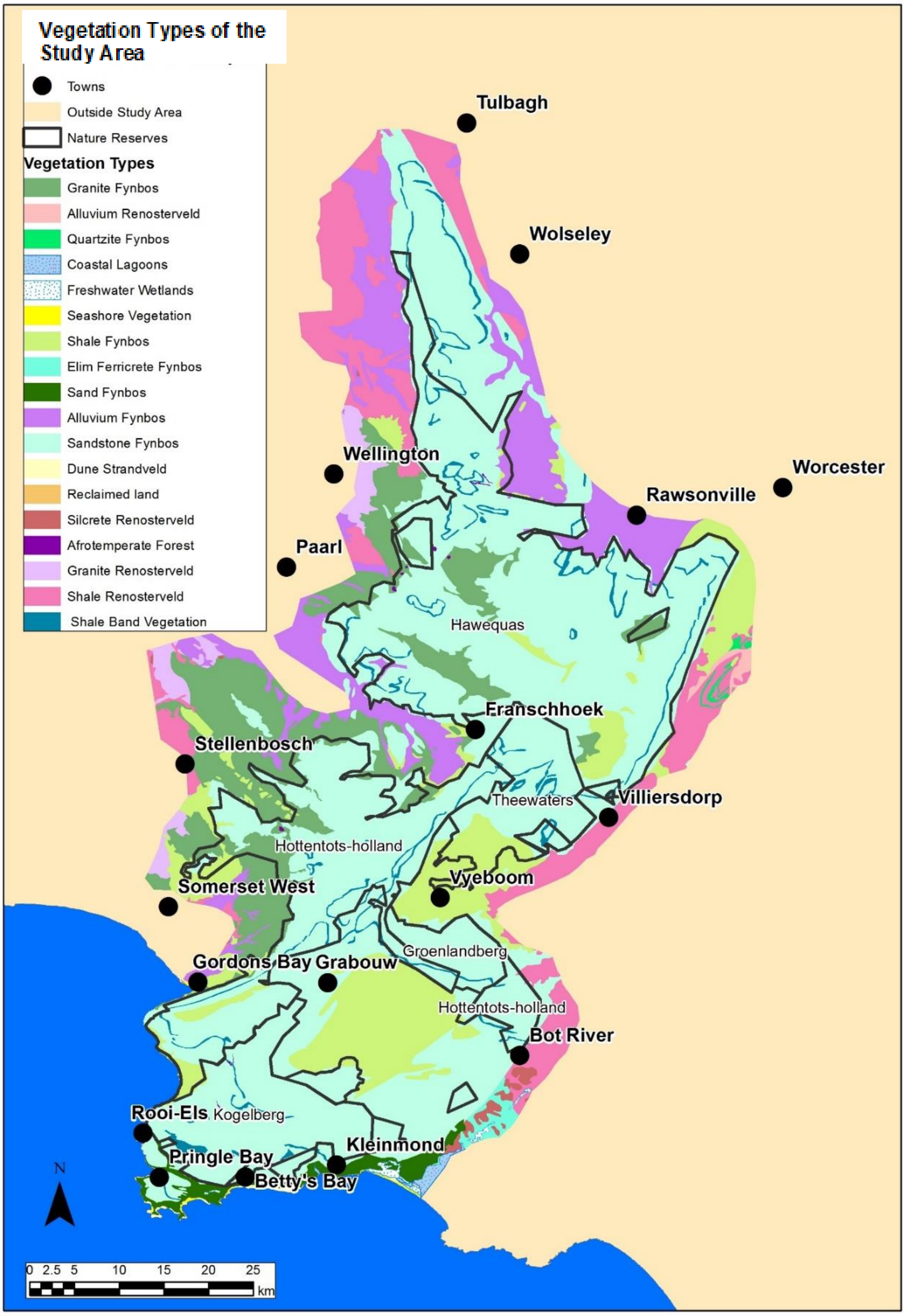


Figure 1.2: Underlying vegetation types in the study area (Mucina et al., 2005).

The study area, within 33.3º - 34.4º S and 18.7º - 19.5ºE, is approximately 3635km^2 in size. It runs from the south of Tulbagh to the coastal southern edge and includes CapeNature's

Hawequas Mountains, the Limietberg, Jonkershoek, Hottentots-Holland and Kogelberg Nature Reserves as well as the City of Cape Town's Helderberg and Steenbras Nature Reserves. It includes sections of the towns of: Wellington, Paarl, Rawsonville, Stellenbosch, Villiersdorp, Somerset West, Bot River and Gordon's Bay. In their entirety it contains the towns of: Grabouw, Franschhoek, Kleinmond, Pringle Bay, Betty's Bay, Rooi Els and Vyeboom, and is bordered by Tulbagh, Wolseley and Worcester. This area makes up the entire KBR and the majority of the CWBR (Pool-Stanvliet & Giliomee, 2013). There are 17 towns across three municipalities, which vary in size and have primary functions including agriculture, tourism and education (Pool-Stanvliet, 2013; Pool-Stanvliet & Giliomee, 2013). Many of these towns have large populations ranging from 125 to 146 526 people and totalling at approximately 726 235 people (Statistics South Africa, 2018). Each has at least one informal settlement. The main agricultural practice is fruit production, primarily grapes, apples and pears. Other farmed products include forestry, cut flowers, game and some domestic livestock (Pool-Stanvliet, 2013; Pool-Stanvliet & Giliomee, 2013). Tourism is an expanding industry, brought about initially by wine production in the area (Stevens, 2003). The study area has six major government dams supplying water to the City of Cape Town and towns in the area.

1.2.1 Human History of the Boland Mountain Complex

The BMC and surrounding areas were home to an abundance of large mammal species in its fertile valleys, prior to human presence (Skead, 1980). Koi and San tribes were the only known humans present from roughly the 1300's (Stevens, 2003). These small human population sizes had seemingly little impact on the natural ecology, but they did ignite and utilise fire (Anderson & O'Farrell, 2012; Barnard, 1992; Stevens, 2003). From the 1650's, with the arrival of the Dutch and other European settlers, many mammal species (initially reported as occurring at high densities in the area) were hunted to localised extinctions (Anderson & O'Farrell, 2012; Skead, 1980). Hunting for food, government rewards, and protection from species perceived as dangerous, initiated large mammal population declines (Skead, 1980). Most large species were eradicated from the landscape and medium-sized mammals were removed from the low-lying, highly developed expanses, and left in isolated mountainous areas that were less accessible to humans (Skead, 1980; Stadler, 2006). The remaining mammals were continuously hunted for various human uses and for pest removal purposes, up until 1974 when legislation was officially launched by Nature Conservation Ordinances (Ordinance 19 of 1974) (South Africa, 2001; Stadler, 2006).

Fire was an abiotic factor that colonialists suppressed from the 1600's until 1968 (Bands, 1977; Southey, 2009). Suppression of fire in the Fynbos biome inhibited the germination of multiple endemic fynbos species, slowed the flow of water in local streams and it made vegetation management difficult because of the spread of invasive plant species and a build-up of native vegetation as fuel (Bands, 1977; Van der Zyl & Kruger, 1975; van Wilgen, 2009; van Wilgen, 2013). In 1968, prescribed burning was put in place by the Department of Forestry (Bands, 1977; van Wilgen, 2009). Many of the development regulations are managed by CapeNature to protect the remaining natural habitats according to the National Environmental Management: Protected Areas Act (NEM:PAA), that was initiated in 1998 (Pool-Stanvliet et al., 2017). Since 2003, CapeNature's Conservation Stewardship, followed by the World Wild Fund for Nature (WWF)'s Conservation Champions programmes were initiated. Both are stewardship agreement programmes that promote sustainable resource use on agricultural properties (see https://www.capenature.co.za/care-for-nature/stewardship/ and https://www.wwf.org.za/our_work/initiatives/conservation_champions.cfm, respectively). These involved environmental management plans, alien clearing and designation of natural habitat patches on properties that are within the BMC that are apart of these programmes.

After the arrival of European settlers, human development in the BMC was based around agriculture due to the fertile valleys (Anderson & O'Farrell, 2012; Mangnall & Crow, 2003; Stevens, 2003). Stellenbosch and Paarl are the second and third oldest towns in South Africa respectively (Stevens, 2003). Many of the road-passes and major highways (N1 and N2) through the study area were tarred after the 1930's, which increased the urban growth of many towns (Mitchell, 2014; Murray, 2015). By the 1970's agriculture had intensified, and it is now widely debated as to how the practice benefits or hinders natural ecosystems (Rudel et al., 2009). The removal of the apartheid government in 1994, was a pivotal year in South African history and had tremendous effects of the physical landscape and the laws that governed the country (O'Laughlin et al., 2013). One may assume that substantial changes took place in all spheres in the 1990's, including those that affected the natural landscape (O'Laughlin et al., 2013).

During the 1980's a few studies within the BMC and surrounding habitats focussed on medium-sized mammals, including the general ecology of leopard and caracal *Caracal caracal* (Norton, 1980; Norton et al., 1986; Novellie et al., 1983). From 2004, research technology advanced locally and with the establishment of conservancies and the Cape Leopard Trust's Boland Project study area in 2010, more research and information was generated on leopards and the other mammals (Frohlich, 2011; Martins, 2010; Radloff, 2008; Swanepoel et al., 2016).

1.2.2 UNESCO Biosphere Reserves

One of UNESCO's most successful projects is the Man and Biosphere (MAB) Program that establishes biosphere reserves with local governments (Ishwaran et al., 2008). Each Biosphere Reserve was designated for reasons specific to its locality, but all have the same goal of benefiting local communities and biodiversity by interlinking the two (Batisse, 1982; Ishwaran et al., 2008). This is achieved through education, information and experience exchange between all stakeholders, thus strengthening conservation legislature and sustainable development in all structural land-use zones (Ishwaran et al., 2008; Price, 2002). Research is a fundamental function, particularly that which incorporates both human and ecological factors for future planning (Batisse, 1982; Ishwaran et al., 2008; Makenzi, 2013). Biosphere reserves allow for findings to be better applied, management changes to be communicated and provide a medium for suggestions to be implemented in multiple aspects of an area (Nguyen et al., 2009).

The South African Biosphere Reserve Program's vision is to be recognized as special landscapes where socio-ecological land management is practiced towards a more sustainable future for all (Pool-Stanvleit, 2013). The two UNESCO biosphere reserves within the BMC were established to maintain the presence and landscape management of the biodiversity hotspot, while balancing ecosystem functions with an array of human development (UNESCO, 1996). The KBR's 1000km^2 (including a marine section) was designated as South Africa's first biosphere reserve in 1998 and it is now the country's smallest (Muller, 2008; Pool-Stanvliet, 2013). The initial focus was to bring about a conservation baseline for assessing threats to the pristine fynbos habitats of the Kogelberg Mountain, specifically after a major fire occurred and a new dam was proposed in the area (Rabie, 2005). The 3220km^2 CWBR was the sixth proclaimed in South Africa in 2007. Its purpose was to integrate the three municipalities (Stellenbosch, Drakenstein and Breede Valley) that share the area's development plans (Pool-Stanvleit, 2013; Pool-Stanvliet & Giliomee, 2013).

Structurally each biosphere reserve has core, buffer and transitional zones (Li et al., 1999). The core zones are defined as the formally protected areas by long-term national law and are often government owned, and primarily managed by CapeNature in the BMC (Makenzi, 2013; Pool-Stanvliet & Giliomee, 2013; Price, 2002). This study area's core zones are mostly mountainous and coastline (Boshoff et al., 2001). Buffer areas, that are usually privately owned, surround and act as matrices to the cores and still function as ecosystems, but are partially developed with lower levels of protection (Makenzi, 2013; UNESCO, 1996). Most valleys and lower slopes that surround the steep core areas form buffers in the BMC, consisting of agriculture and forestry (Boshoff et al., 2001). The transitional zones are located outside the buffer zones and include a gradient of human functions (Makenzi, 2013). In the BMC, these areas contain intensive agriculture, industrial processes and a diversity of urban activities (Ishwaran et al., 2008; Makenzi, 2013). The aim of the transitional zones, as part of the UNESCO Biosphere Reserves, are to harbour sustainable human activities and to integrate local communities within conservation (Ishwaran et al., 2008). The biosphere reserves' three zones allow for a structure of comparison to detect changes to human development and nature, with a strong stakeholder input and acts of adaptive planning (Batisse, 1982; Folke et al., 2005). The demarcation of these zones by the biosphere reserves is therefore important in maintaining a balance in the landscape.

1.2.3 Major mountain zones

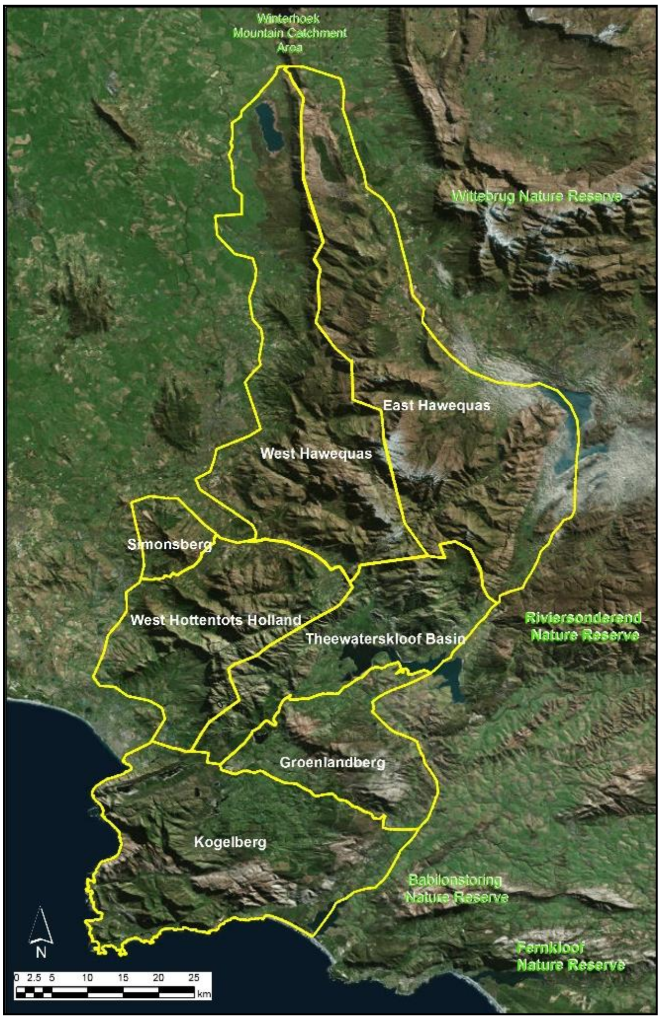


Figure 1.3: Satellite image of the study area with the outlines of the seven defined major mountain zones of the Boland Mountain Complex indicated in yellow

The study area was subdivided into seven major mountain zones (MZ) that reflect human management divisions and potentially act as barriers to mammal populations (Figure. 1.3). These were primarily divided based on how sample sites clustered in nearby locations to one another (see Chapter 3 of this study) and were outlined in ArcMap 10.4.1. (ESRI, 2015). The zones were further defined with apparent barriers such as roads, non-natural land-covers, steep mountain slopes and along the demarcated borders of municipalities and biosphere reserves. The study's strong representation of human development makes these an appropriate way to implement future management and relatable planning with landowners. Kogelberg MZ (691.04km^2), Groenlandberg MZ (296.1km^2), Theewaterskloof Basin MZ (396.24km^2), West Hottentots-Holland MZ (453.32km^2), Simonsberg MZ (77.23km^2), West Hawequas MZ (797.1km^2) and East Hawequas MZ (920.42km^2) were the names used to describe these sections. Comparisons were made between the CWBR and KBR which overlap with these MZ borders.

1.3 Human development

With many towns of varying population sizes, growing economies and road infrastructure, the BMC has incurred large degrees of human development and expansion overtime. Jacobson et al. (2016) identified habitat destruction and fragmentation as prevalent threats to leopards globally, and in South Africa it is expected that many of the leopards mammalian prey species encounter the same threats. Land-cover changes, habitat-loss, fragmentation and urbanisation due to human development are some of the biggest threats globally to biodiversity and have already caused many localised extinctions since human expansion in the Cape (Boshoff et al., 2002; Brooks et al., 2002; Hansen & DeFries, 2007; Rouget et al., 2003). Norton et al. (1986) and others, previously described how human development covered most of the lowland areas of the BMC, and resulted in a decrease in faunal abundance (Boshoff et al., 2001; Mangnall & Crowe, 2003; Mann et al., 2019; Rouget et al., 2003). These extinction risks increased in larger mammals, and it is probable that further habitat loss next affected medium-sized mammals (Boshoff et al., 2001; Cardillo et al., 2005). Both the physical quantity of habitat available and what the habitat consists of, are important to mammalian species' survival (Bender, Contreras & Fahrig, 1998; Eigenbrod et al., 2008; Schneider, 2001).

Buffer zones are the forefront of where human development has the most direct and obvious effects on natural habitats and core zones (Kintz et al., 2006). Agricultural expansion likely introduces some of the greatest negative impacts and threats of habitat destruction to mammals in the BMC, as it has done in the past (Rouget et al., 2003). The latter authors predicted that by 2023, a further 30% of the then remaining 74% of the original Fynbos biome would have been replaced by unnatural land-covers. Viniculture is one of the land-uses that underwent great expansion since the nineties and Fairbanks et al. (2004) predicted it would cause the greatest encroachment into natural habitats within the study area. More recently Hannah et al. (2013) also highlighted concern for viniculture expansion and shifts into natural vegetation in the region, driven by climate change. One of the only land-use studies to look at predicted changes in the Western Cape, highlighted Stellenbosch as having one of the fastest urban increases encroaching on surrounding agriculture, forestry and natural habitat areas (Tizora et al., 2016). Built-up areas do not expand into large areas of habitat, but have covered small habitats like those near Grabouw within the BMC, which Rouget et al. (2003) described as irreplaceable (Cowling, 1992). Roads are prominent drivers of habitat fragmentation that allow humans to penetrate the surrounding landscape (Alphan, 2017; Sauvajot et al., 1998). Donaldson et al. (2012) also flagged many towns in the study area as having high urban development rates. Together with the booming agricultural activities, risks to remaining mammalian habitat is of great conservation concern.

Calculating the size of land-cover change is the initial step in quantifying the impacts of human development expansions on local habitats and is required for future planning (Tizora et al., 2018). Land-cover changes with time, pose the same general threats to biodiversity globally, yet can differ vastly in scale and precise impacts (Hansen & DeFries, 2007; Harrison & Bruna, 1999; Wilcove, et al., 1998). Many studies have quantified land-cover size changes across the planet (Harrison & Bruna, 1999; Rouget et al., 2003). Once a general quantification is calculated and with a strong understanding of local species' ecologies, habitat loss and impacts on the ecosystem can be assessed (Bender et al., 1998; Eigenbrod et al., 2008; Lambert et al., 2006; Schneider, 2001). The total area of habitat can affect mammal presence, population size, species richness and survival rate (Bender et al., 1998; Schneider, 2001; Silva et al., 2005). If habitat loss and availability is correctly understood, it has the potential to be efficiently managed to sustain species and combat many negative impacts of human encroachment (Schneider, 2001).

Tizora et al.'s (2016) conference paper of the Western Cape's land-cover changes, provides a good overview of the landscape on a broad scale, which is vital for spatial planning by provincial and national governments. However, few studies have looked at land-cover changes at a smaller scale or with specific reference to localised ecological functioning, and none with a specific focus on the BMC (Geyer et al., 2011; Halpern & Meadows, 2013; Heijnis et al., 1999). When assessing habitat availability for flora, the size of a habitat is much less important than its contents, but to fauna it is both the space available and the quality of the habitat which are important (Lambert et al., 2006). The BMC is used because of its localised ecology of a functional-sized landscape (assumed to house multiple populations of most study species) and is assessed according to the loss of medium-sized mammalian habitat.

1.4 Fires

Since early human history, fire has been a life-changing factor that can however also cause mass destruction (Archibald et al., 2012). When controlled it has pushed human development and is currently utilised in everyday life (Archibald et al., 2012). In fire-dependant biomes like fynbos, fire management is thus a major priority and man's understanding of it has changed over the years, leading to unnatural regimes and more recently an increased frequency and size of wildfires (Archibald et al., 2012; Gumbi, 2011; Sturtevant et al., 2009). Syphard et al. (2009) found that across all Mediterranean climate systems, such as that of the Cape, humans affect the way that wildfires burn. Within the study area humans are the source of most fire ignitions and drive increased fire frequency in the Hottentots-Holland Nature Reserve (Bond & van Wilgen, 1996; van Wilgen et al., 2010). Human population density is thus the primary factor influencing this ignition source and fire frequency (Syphard et al., 2009). Fire management, in many cases, aims to suppress fires in order to lower mortal and financial risks (Kraaij et al., 2011). However, this leads to an accumulated fuel-load and consequently greater fire sizes and increased fire frequencies (Bands, 1977; Kraaij et al., 2011; Minnich & Chou, 1997). Additionally, numerous invasive plant species that were introduced by humans create high fuel-loads, resulting in more intense, severe fires and seeds that are then further dispersed by natural fires (Kraaij et al., 2011; Mack & D'Antonia, 1998; Syphard, et al., 2009).

With the localised extinction of large grazers like Cape buffalo *Syncerus caffer*, African elephant *Loxodonta africana* and quagga *Equus quagga quagga* in most of the Cape region, fire is the remaining natural disturbance (Skead, 1980). It has played a vital role in the evolution of fynbos and renosterveld vegetation types (Forsyth & van Wilgen, 2008). As a key abiotic factor in the ecology of the Fynbos biome most of the flora and fauna in the region are adapted to a specific regime (Forsyth & van Wilgen, 2008). This regular burning controls the vegetation structure and triggers reproduction in many plant species, which then drives the evolution of new survival traits and with time, new species (Bond et al., 2005; Keeley et al., 2011). Both extremes of too frequent or not frequent enough fires threaten the ecosystem, may prevent vegetation from reproducing and can cause a change in the physical structure of the vegetation and its ability to support faunal biodiversity and drive genetic diversity (Chan et al., 2011; Keeley et al., 2011; Van Wilgen et al., 2010). This may alter what habitats are available over time and potentially change the abundances of medium-sized mammals, which according to Cowling et al, (1996) drive vertebrate diversity (Barro & Conard, 1991; Bigalke & Willan, 1984).

Although the role of fire regimes is important throughout the Fynbos biome, the period lengths between burns vary greatly between vegetation types, locations and species (Schutte et al., 1995; Van Wilgen, 2009). The nearest fire regime studies to the BMC conclude that the Fynbos biome has an average burning cycle of every 15 years (van Wilgen et al., 2010; van Wilgen & Scott, 2001; Vlok & Yeaton, 2000). Although factors indicated that fires still occurred at relatively sustainable rates, by 2006, Gumbi (2011) found that the fire frequency of the Kogelberg Nature Reserve had increased since 1980 and was burning more frequently than optimum. Fire ecology research has focussed on broad-scale spatial and temporal patterns of fires, the ecological impacts of fire on vegetation and in a few incidences, the effects of fire on smaller mammals and other fauna that play key roles in the dispersal of many fynbos species (Auld & Denham, 2001; Forsyth & van Wilgen, 2008; Gumbi, 2011; van Hensbergen et al., 1992; van Wilgen et al., 2010; Willan & Bigalke, 1981).

Little focus on the Fynbos biome's fire ecology has been how fires affect fauna, particularly medium-sized mammals which are a fundamental part of the biome's biodiversity (Rebelo et al., 2019). Southey (2009) suggested that fire's effects on mammals must be monitored and it is important to understand fire regimes and related ecological implications with reference to mammals' habitat availability (Pausas & Parr, 2018). Bigalke & Willan (1984) stated that larger mammals are often able to escape fires and avoid injury/death but are likely affected by habitat damage. Mammals are often forced to flee into neighbouring, possibly sub-optimal habitats and therefore an assessment of spatial fire data across both core and human-developed buffers is vital (Legge et al., 2008). In many ecosystems there are recorded situations where recently burnt areas of habitat benefit some mammal species (Eby et al., 2014; Jaffe & Isabell, 2009; Pausas & Parr, 2018). A spatial study, of a size that incorporates the majority of influences and landscape patterns, is required to understand whether sustainable mammalian habitat remains post-fire (Gumbi, 2011; Richardson et al., 1994). Fire-management strategies that are initiated in the Fynbos biome should include an analysis of fire records dating as far back as 30-years and further. This is the maximum period between burning that particular species and patches of fynbos can withstand until senescence becomes a risk in parts of the Fynbos biome (Forsyth & van Wilgen, 2008; van Wilgen, 2009; Vlok & Yeaton, 2000). Understanding of fire regimes in relation to human land-covers is relevant for stakeholders to understand and initiate successful management strategies (Syphard et al., 2007).

1.5 Medium-sized mammals

Fossil records and historic accounts indicate that the Western Cape has lost many of its large, naturally occurring mammalian species in most PAs and across unprotected land-covers (Birss, 2017; Kerley et al., 2003; Skead, 1980). Blue antelope *Hippotragus leucophaeus,* quagga *Equus quagga quagga*, Cape warthog *Phacocheorus aethiopicus aethiopicus* and Cape lion *Panthera leo melanochaitus* are species and subspecies in the Western Cape, that are now extinct (Birss, 2017). Large and medium-sized mammals, such as the African elephant, black rhinoceros *Diceros bicornis*, bontebok *Damaliscus pygargus,* African lion *Panthera leo*, brown hyena *Hyaena brunnea*, cheetah *Acinonyx jubatus*, serval *Leptailurus serval* and African wild dog *Lycaon pictus*, no longer occur in the study area and only remain in a few small, protected patches of their once expansive range (Birss, 2017; Boshoff et al., 2001; Skead, 1980; Stuart et al., 2003). Many remaining large and small mammals have undergone severe distribution loss and population declines in the remaining habitats (Boshoff et al., 2001). Small populations of some large mammals have been reintroduced into various reserves and other PAs, but for many species this is not always feasible due to reduced habitat size, habitat type and altered migration corridors they require (Birss, 2017).

The Western Cape is often thought to have a low abundance and diversity of naturally occurring mammalian species. Population sizes of many species have been shown to be lower in the Fynbos biome, specifically when compared to the rest of South Africa (Klein, 1983; Radloff et al., 2010). Mann et al. (2019) described the lower diversity of mammal species within the BMC (because of greater expanses of agriculture over the lowland habitats) than in the Little Karoo. That said, the diversity of medium-sized mammals remains relatively high and contributes to the area being a biodiversity hotspot (Kerley, et al. 2003). Most studies on mammals in PAs within the BMC have focused on the smaller, seed dispersing species, such as gerbils and mice (Bond et al., 1980; Fraser, 1990). Other studies have a stronger focus on the large mammals and least of all the medium-sized mammals (Boshoff et al., 2002; Kerley et al., 2003; Radloff, 2008; Smith et al., 2007).

Due to their smaller sizes, habits and low impacts on daily human lives, medium-sized mammal species have limited historic and current data on their abundances (O'Connell et al., 2006). Mammal species that have undergone local extinctions and severe declines in the Fynbos biome tend to require lowland habitats (Boshoff et al., 2001; Skead, 1980). Lowland fynbos and renosterveld habitats are often the most productive areas for agriculture and/or forestry and are therefore more convenient for human settlement (Boshoff et al., 2001; Fairbanks et al., 2004). Species like the African striped weasel *Poecilogale albinucha* were once considered locally extinct in patches within the Fynbos Biome and BMC but have been observed and recorded in recent years (Birss, 2017; Kerley et al., 2003). Typically, the natural vegetation supports fewer grazers and consists of more mixed feeders that are small to medium-sized, due to the evolution of limited grasslands and a low vegetation productivity (Faith, 2011; Radloff et al., 2010).

This study considers the threats to the medium-sized mammalian species. These species include the leopard's primary prey-base and other mammals that can and do make opportunistic prey options. With this perspective, the study species are therefore 29 medium-sized mammals. All 29 of the study species (Appendix 1.1), have been captured and identified on camera traps in recent years while attempting to photograph individual leopards (Wilkinson, personal communication, Cape Leopard Trust, 2018). Of the medium-sized mammalian study species, camera traps recorded African striped weasel as the smallest study species and the largest were grey rhebok *Pelea capreolus* and leopard *Panthera pardus pardus* (Stuart & Stuart, 2015). Only the Cape grysbok *Raphicerus melanotis* can be considered an endemic medium-sized mammal species to the Fynbos biome (Apps, 2012). Of the 29 study species, 26 were native to the area.

Roles fulfilled by a diversity of mammals include that of top predators, meso-predators, insectivores, omnivores, herbivores and invasive mammals, which make up habitat's ecological food web with various associations with each other or with various biotic variables (Duffy et al., 2007; Richie & Johnston, 2009; Sergio et al., 2008). Top predators are often the least diverse, have various top-down cascades on all other classes, experience few sources of competition and are rarely preyed upon by other species in an ecosystem (Elmhagen et al., 2010; Sergio et al., 2008). A low predator presence in an ecosystem may negatively affect other species' richness and diversity and misalign the balance between lower trophic levels and drive a process called mesopredator release (Crookes & Soulé, 1999; Fonseca & Robinson, 1989; Letnic et al., 2009). Meso-predators are often smaller, more diverse carnivores in an ecosystem, whose populations can be influenced by both their prey and top predators (Ritchie & Johnston, 2009; Roemer et al., 2009). Omnivores have both bottom-up and top-down effects and may act as competitors and predators on multiple trophic levels, thus having a broader influence on more species and biotic factors in an ecosystem (Duffy et al., 2007; Morris, 2005). Insectivores often have strong influences on biotic factors, such as the presence of burrows, behaviour of insects as well as the dispersal and germination of seeds (which is important after fire) (Quinn, 1986; Shiponeni & Milton, 2005). Herbivores are disturbance agents (often acting with fire) to floral structures and biodiversity and generally make good prey sources for local predators (Hayward et al., 2006; Palmer & Fairall, 1988; Radloff, 2008; Shiponeni & Milton, 2005; Stuart & Stuart 2015).

1.5.1 Top Predators

Leopards are considered to be the largest, natural apex predator in the BMC's ecosystems (Martins & Martins, 2006). Their opportunistic feeding behaviour is important to consider when attempting to understand the ecology and abundance of the various mammalian prey species available to them, and to ensure that most mammals can persist in extremely altered habitats (Hayward et al., 2006). Leopards, as apex predators, may influence many species' behaviours and drive trophic cascades (Crookes & Soulé, 1999; Sinclair et al., 2010; Ripple & Beschta, 2004). Their most common prey items in the BMC are from three different herbivorous orders of taxa; klipspringers *Oreotragus oreotragus*, grysbok, rock hyraxes *Procavia capensis*, and Cape porcupines *Hystrix africaeaustralis,* which make up 80% of their diet (Mann et al., 2019). Other less common species contained in their scat samples most recently included chacma baboons *Papio ursinus,* hares *Lepus spp.*, common duiker *Sylvicapra grimmia,* African wild cat *Felis silvestris lybica*, livestock, grey rhebok and caracal (Mann et al., 2019). Prey species that were consumed at lower frequencies or were found in historic studies in the BMC included: feral pigs *Sus scrofa*, Cape grey mongoose *Galerella pulverulenta*, African striped weasel and various rodents (Mann et al., 2019; Norton et al., 1986). Other study species found in their diet in other Fynbos biome ranges include mongoose, aardwolf *Proteles cristata*, and genets *Genetta spp.* (Frohlich, 2011; Hayward et al., 2006; Mann et al., 2019; Martins, 2010; Norton et al., 1986; Rautenbach, 2010). In the Fynbos biome the leopards home ranges have spanned up to 910km^2, but individual females can persist within a 100km^2 home range (Boshoff et al., 2002; Lindsay, 2008). Males usually hold a much larger home range than females, which can encompass multiple female ranges (Skinner & Smithers, 1990). Local leopard populations have larger home ranges than in other regions, which may be to increase the chance of encountering relatively smaller-sized prey that seem to occur at lower abundances (Baily, 1993; Marker & Dickman, 2005; Odden & Wegge, 2005).

Caracal are the second largest felid species that remain in the Fynbos biome, with an average mass of 12.8kg (Pringle & Pringle, 1979). They are generalist species occurring in almost all habitats with an extremely broad prey choice (Melville et al., 2004; Palmer & Fairall, 1988). Mammals, including rodents, rock hyraxes, hares, grey rhebok, common duiker, grysbok, klipspringer, smaller carnivores (such as mongoose, African wild cat, domestic cats *Felis catus*), and livestock, are commonly preferred prey items (Grobler, 1981; Melville et al., 2004; Mukherjee et al., 2004; Palmer & Fairall, 1988). Birds, reptiles, amphibians, fish and invertebrates have also been documented in the caracal's diet (Palmer & Fairall, 1988). They are often involved in human-wildlife conflict due to preying on livestock and domestic pets (Bothma, 2012; Braczkowski et al., 2012a). Thus, caracal adapt well to natural or disturbed terrain (Palmer & Fairall, 1988). Females appear to have smaller home ranges than males and together a pair experiences some overlap in home ranges. Boshoff et al. (2002) concluded that they require an average minimum of 24.6km^2 per caracal.

Both leopard and caracal represent the low diversity of natural top predators in the BMC, and they both prey opportunistically on a broad, overlapping variety of prey species (Frohlich, 2011; Grobler, 1981; Hayward et al., 2006; Martins et al., 2011; Melville et al., 2004; Mukherjee et al., 2004; Norton et al., 1986; Palmer & Fairall, 1988; Rautenbach, 2010). In most studied populations, caracal diet consists primarily of rodents and many birds and smaller faunal species (Grobler, 1981; Lloyd & Millar, 1981; Matthews, 2002; Melville et al., 2004; Stuart & Stuart, 2015). Leopards in the Western Cape generally select larger prey species and additionally, caracal remains have been found in their diet (Mann et al., 2019; Norton et al., 1986). Both species are solitarily, but Norton & Lawson (1985) found that, unlike leopards, caracal generally do not exceed 600m in elevation within the study area, which would reduce competition for resources (Caro & Stoner, 2004; Stein & Hayssen, 2013). Caracal are in general, more commonly sighted on agricultural land, in residential areas and close to urban cities than leopards (Bothma, 2012).

1.5.2 Meso-predators

The term "meso-predator" is defined either by body mass of intermediate predators (Buskirk, 1999), or according to their role in the ecosystem (Prugh et al., 2008; Roemer et al., 2009). For this study, like Prugh et al. (2008), "meso-predators" refer to those carnivores that are affected by the apex predators and exist as various and often diverse taxa, shapes and sizes (Ritchie & Johnston, 2009; Roemer et al., 2009). Whether a species is classified as a meso-predator depends on the ecosystem the specific population occurs in (Prugh et al., 2008). For example, where lions and spotted hyenas are present, African wild dog take the role of meso-predators, but without lion or spotted hyena populations, wild dogs take on the role of apex predator (Creel, 2001; Ritchie & Johnston, 2009). Their roles include controlling population sizes of small mammals and other vertebrates, across many taxa (Kamler et al., 2013), and they often aid in the consumption and decomposition of carcasses (DeVault et al., 2011). The guilds compete in multiple different ways in an ecosystem and may be opportunistically preyed upon by larger predators (Caro & Stoner, 2004). Although diverse, meso-predators are less researched than the larger or apex predators (Kamler et al., 2013; Roemer et al., 2009; Thompson & Gese, 2007).

The African wildcat has a wide distribution but is infrequently seen and has seldom been studied (Herbst & Mills, 2010). They require a substantial amount of bush for cover (Stuart et al., 2013). Their scat largely contains rodents, birds, invertebrates, reptiles and a small amount of plant matter (possibly secondary ingestion) which can therefore result in competition with caracal (Palmer & Fairall, 1988; Stuart et al., 2003; Stuart & Stuart, 2015). More seldom, rock hyrax, hares, poultry and occasionally the lambs of small antelope and domestic livestock are also consumed (Stuart et al., 2013; Stuart & Stuart, 2015). The African wildcat has been recorded in the diet of both leopard and caracal and these hunting incidents have also been visually recorded (Grobler, 1981; Macdonald et al., 2010; Melville et al., 2004). In urban areas domestic cats compete with African wildcat, and potential interbreeding can threaten the species' gene pool. They are also often considered to be pests and are thus often eradicated (Conradie & Piesse, 2013; Le Roux et al., 2015; Palmer & Fairall, 1988). Their average minimum home range is 1.25km^2 and like most felids, the males' home ranges can contain up to four females' home ranges (Boshoff et al., 2002; Herbst & Mills, 2010).

Cape fox *Vulpus chama* are widespread, generally utilising grasslands and some scrublands, while often living nocturnally in pairs (Kamler & Macdonald, 2014; Nel, 1984; Skinner & Chimimba, 2005; Stuart & Stuart, 2015). They are carnivorous and have been recorded feeding on new born domestic lambs, but in general their diet consists of hares, rodents, birds, carrion, arthropods, reptiles and grass (Kamler et al., 2012; Nel, 1984; Stuart & Stuart, 2015). The Cape fox' distribution ranges across Southern Africa have shrunk, and they are often mistakenly perceived as pests to livestock farmers, where they are then persecuted (via hunting or poison) (Kamler & Macdonald, 2014; Stuart & Stuart, 2015). On other properties they are mistakenly persecuted as by-catch when attempting to eradicate true pest species, such as jackal species. (Conradie & Piesse, 2013). Generally however, the species do a good service in agriculture as they assist with rodent removal (Kamler & Macdonald, 2014; Stuart & Stuart, 2015). Their distribution is influenced by the presence or absence of jackal *Canis spp.*, and Cape fox remains have been found in the scat of leopard, caracal and honey badgers *Mellivora capensis* (Hayward et al., 2006; Kamler et al., 2013; Mills, 1984). Although similar in appearance, many aspects of their ecology differ from the predominantly insectivorous bat-eared foxes *Otocyon megalotis*. They experience more competition from other meso-predators (Kamler et al., 2012; Stuart et al., 2003) and according to Boshoff et al. (2002) a pair of Cape fox can inhabit 7.5km^2 or an individual an area of 3.25km^2.

Cape clawless otters *Aonyx capensis* are a species that encounter few predators anywhere in their range (Somers & Nel, 2013). The species are carnivorous and feed in and near both fresh and ocean water, with habitat preference being thick riparian zones or bush for shelter (Larivière 2001). Prey species include waterfowl and other birds, eggs, rodents, reptiles, amphibians and invertebrates (Somers & Nel, 2013). The KBR's coastline is home to many individuals (Rabie, 2005). Water surface area is a more important factor than land surface area in Cape clawless otters' home range size (Kruuk & Moorhouse, 1991; Somers & Nel, 2004), but Somers and Nel (2004) found smaller portions of agricultural dams are utilised than natural water sources. They have a crepuscular activity pattern (Larivière 2001). Exact home ranges of otters depend on food and water availability and individuals have been found to utilise a minimum of 2km of coastline (van der Zee, 1981) and from 0.05 to 10.63km^2 of water surface area (Jacques et al., 2015). Due to their aggression towards other species including domestic animals, and their habit of preying on poultry and farmed and garden fish, human-wildlife conflict does occur (Stuart & Stuart, 2015). That said, habitat disruption, loss of riparian zones and the pollution of waterways negatively affects the species (Stuart & Stuart, 2015).

African striped weasel and striped polecat *Ictonyx striatus* are similar looking members of the musilidae family. The striped polecat has been more studied than the African striped weasel which was previously described as being extinct in the BMC (Stuart & Stuart, 2013). The polecat's local recorded predators are caracal, leopard and raptors (Stuart & Stuart, 2013). Both species are nocturnal carnivores, with weasels specializing in rodents, whilst the polecat is more of a generalist, feeding on rodents, reptiles, birds, eggs, amphibians, invertebrates and possibly poultry (Stuart & Stuart, 2013). Polecats dig for insects and other prey, but weasels create their own burrows by digging after mice, which plays an important role in soil and vegetation structure (Stuart & Stuart, 2013; Stuart & Stuart, 2015). Polecats are ranked on the IUCN list as "least concern", while weasels are listed as "threatened". Both species are regularly recorded as roadkill, are often attacked by domestic dogs *Canis lupus familiaris*, and additionally, loss of suitable habitat is a growing threat (Birss, 2017; Stuart & Stuart, 2013; Stuart & Stuart, 2015). Few studies have specifically focussed on threats to these species in the Fynbos biome and to date, home range sizes have not been determined. Recently, striped weasels have once again been found in the BMC, in seemingly stable populations, likely due to high rodent availability on farms (Birss, 2017).

Honey badgers have not been recorded as prey for any of the study species. They make fierce competitors and are well known predators and scavengers (Begg et al., 2001). Their main prey species include rodents, reptiles, some smaller mammals (hares, mongooses, bat-eared fox, Cape fox and small antelope), birds, eggs, and invertebrates and their larvae (particularly bee's) (Begg et al., 2001; Kruuk & Mills, 1983). They hunt for insect larvae by digging new burrows daily, which aerates the soil (Begg et al., 2001). Honey badgers occur across multiple habitats in low densities, with large ranges of a minimum of 50km^2, which has contributed to the limited research conducted on them (Boshoff et al., 2002; Do Linh San et al., 2016). Their IUCN status is “least concern” and human-wildlife conflict is one of their greatest threats, as they are known to raid bins in campsites, destroy farmed beehives, and they hunt poultry, goats and sheep (Apps, 2012; Do Linh San et al., 2016; Stuart & Stuart, 2015).

Three species of mongoose occur in the study area, namely: the large grey mongoose *Herpestes ichneumon*, Cape grey mongoose and water mongoose *Atilax paludinosus*. Each of the three species fill a different niche in the ecosystems they occupy and thus exhibit prevalent habitat partitioning (Avenant & Nel, 1997). The three species’ diets include rodents, birds, reptiles, eggs, amphibians, fish, invertebrates and fruit (Cavallini & Nel, 1990; Stuart & Stuart 2015). The diet of the water mongoose includes more aquatic animals, which leads to competition with the Cape clawless otter (Avenant & Nel, 1997; Purves et al., 1994). Large grey mongoose also predate on smaller mammals, for example the lambs of Cape grysbok (Stuart & Stuart 2015). Cape grey and water mongoose will scavenge on carrion and in human rubbish (Stuart & Stuart, 2015). All three species experience conflict with humans as they raid poultry and fish farms (Stuart & Stuart, 2015). Leopard, caracal, honey badgers as well as domestic animals prey on mongoose (Braczkowski et al., 2012a; Braczkowski et al., 2012b; Hayward et al., 2006; Kruuk & Mills, 1983; Rautenbach, 2010). Additionally, mongoose have been reported as roadkill and have been attacked by domestic dogs (Bullock et al., 2011; Serieys et al., 2019).

Water mongoose are crepuscular and live solitarily or in pairs throughout Africa, close to water sources (Baker, 1997; Ray, 1997; Stuart & Stuart 2015). Large grey mongoose live solitarily, in pairs or as family groups and are most active mid-morning and mid-afternoon (Palomares & Delibes, 1993a). Large grey mongoose are often located near riparian undergrowth but also occur in a broader range of grassland habitats (Palomares & Delibes, 1993b). Large grey mongoose expanded their range into the BMC of the Cape in the 1950’s (Palomares, 2013; Palomares & Delibes, 1993b). The Cape grey mongoose is a diurnal generalist, occurring in a wide range of dense bushy habitats, but is endemic to Southern Africa (Doh Linh San & Cavallini, 2015). Cape grey mongoose’s home ranges can be less than 1km^2, large grey mongoose home ranges extend up to 4.5km^2 and water mongoose require a minimum home range of 1.5km^2 (Apps, 2012; Cavallini & Nel, 1995).

The large spotted genet *Genetta tigrina* and small spotted genet *Genetta genetta* species are poorly studied throughout their ranges in Africa, Europe and West Asia (Admasuet al., 2004; Gaubert et al., 2015). Both species inhabit fynbos with the large spotted genet showing preference to more forested, bushy areas where more water is present. The small spotted genet occurs in more sparse, arid regions (Stuart & Stuart, 2015). In South Africa both genet species require 1km^2 or less as home ranges (Apps, 2012; Stuart & Stuart, 2015). Genets are primarily nocturnal and occasionally nest in human structures such as rooves or garages (Apps, 2012). Small mammals (bats, rodents and moles), amphibians, reptiles, invertebrates, birds, eggs, fruit and occasionally poultry make up their diet (Gaubert et al., 2015; Stuart & Stuart, 2015). Leopard and caracal do occasionally prey on genets, and other noted threats include road-side collisions, predation by domestic species and human wildlife conflict (Braczkowski et al., 2012a; Braczkowski et al., 2012b; Hayward et al., 2006; Lloyd & Millar, 1981; Serieys et al., 2019).

1.5.3 Insectivores and omnivores

Baboons are the only primate species to occur in the southwestern Cape. Baboons are extremely adaptive, living anywhere they can access water, rest in trees or cliffs and forage for food (Hoffman & O'Riain, 2012). The species are omnivorous, foraging for mostly bulbs, grass and fruit, and hunting for invertebrates, eggs, frogs, fish, birds and smaller mammals (Apps, 2012; Stuart & Stuart, 2015). Their foraging methods are extensive, playing important roles in seed dispersal and germination, and acting as a natural disturbance for habitat change (Bronikowski & Altmann, 1996). Their adaptive behaviour and destructive foraging methods make them extremely susceptible to human conflict, costing farmers high economic losses (Fehlmann et al., 2017). Leopard are responsible for 30% of baboon deaths in most African populations, affecting troop movements and habits, but male baboons have also been recorded killing a leopard in defence (Busse, 1980; Cowlishaw, 1994). An individual adult baboon may only require 0.43km^2 of habitat, but large troops may require as much as 34km^2 (Boshoff et al., 2002).

Porcupines are omnivorous rodents with most of their nutrients coming from organic matter (including those with high toxicity) and insects (Barthelmess, 2006; Bragg & Child, 2016; Happold, 2013). Their roles in ecosystems are as primary consumers, seed dispersers and aiding germination, ecosystem engineers, drivers of habitat disturbance and prey for leopards and often other carnivores that may scavenge on their remains (Barthelmess, 2006; Happold, 2013). Porcupines will reuse other species burrows, and their own are occasionally reused by other mammalian species including fox, hare, aardvark and aardwolf (Happold, 2013). Bothma et al. (1997) described a leopard giving birth in these shelters in the Kalahari (Stuart & Stuart, 2015). Porcupines generally require a minimum home range of 0.5km^2 (Kerley et al., 2003). Their habits of gnawing through pvc irrigation piping, burrowing under and collapsing infrastructure and ring barking trees (leaving them exposed to mortality risks like micro fungi and fire), often make them pests to agriculture and in suburban gardens (Cassola, 2016; Happold, 2013). They are often hunted for their meat and for use of their quills in traditional medicine and to be sold as souvenirs (Chevallier & Ashton, 2006; Stuart & Stuart, 2015).

Bushpig *Potamochoerus larvatus* sightings have recently been reported and captured on camera traps in the BMC (Wilkinson, personal communication, 2018). This species of pig is not native to the area but most recently two populations of the species have been recorded north east and south east of the BMC (Stuart & Stuart, 2015; Venter et al., 2016). They are nocturnal and prefer dense bush or forest and areas of higher rainfall (Breytenbach & Skinner, 1982). As small to larger family groups, they require large ranges and can travel up to 4km during a foraging session (Apps, 2012). Their diet consists of vegetation, particularly bulbs, roots and fruit, but includes many invertebrates, frogs, reptiles and carrion of mammals (Breytenbach & Skinner, 1982; Stuart & Stuart, 2015). For larger predators such as leopard, they are an ideal prey species (Apps, 2012; Braczkowski et al., 2012b). Human threats to this species include hunting (as pest removal or for bushmeat) and habitat fragmentation (Stuart & Stuart, 2015).

Aardvark *Orycteropus afer,* aardwolf and bat-eared fox are three insectivorous species that occur in the BMC. The distribution of insects, specifically ants and termites are strong determinants of these insectivores' presence (Cooper & Skinner, 1979; Skead, 1980; Smith & Smith, 2015). In the Karoo, insectivorous mammals have high seed dispersal and germination success rates by depositing them in their burrows after ingesting the seeds while consuming termite or ant nests (Shiponeni & Milton, 2005). Although this role has not been recorded in the Fynbos biome, it is probable that insectivores contribute in similar ways in the BMC. Due to their specialisation in ants and termites, these species are also susceptible to secondary poisoning from insecticides and/or pesticides and loss of some insect species can cause localised extinctions (Smith & Smith, 2015).

The aardvark is a nocturnal species, with almost 100% of their diet made up of termites and ants and infrequently other insects, mice, fungi and fruit (Apps, 2012). Their digging of large burrows (which are often reused by other study species, such as porcupines, badgers and foxes) and excavations into termite ant nests make them excellent ecosystem engineers (Apps, 2012; Whittington-Jones et al., 2011). This aerates and changes nutrient composition of the soil and structure of the habitat (Whittington-Jones et al., 2011). Their general home range is estimated at 37.5km^2 (Boshoff et al., 2002). Conflict with humans can arise when excavations affect infrastructure on farms and roads, resulting in physical and financial damage (Stuart & Stuart, 2015). Aardvark meat is consumed and used as traditional medicine by certain ethnic cultures, but they are rarely preyed upon by any carnivores (Smith & Smith, 2015; Taylor & Lehmann, 2015).

Aardwolf occur in most habitats, showing a preference for scrub vegetation (Smith & Smith, 2015). Their diet is solely insectivorous, consisting primarily of ants and termites (Cooper & Skinner, 1979; Smith & Smith, 2015). In most habitats they are rare but of "least concern" on the IUCN's red data list (Green, 2015). Their home ranges of 10km^2 are shared by both males and females (Boshoff et al., 2002). Habitat loss, road collisions and attacks by domestic dogs are potential threats to the species (Smith & Smith, 2015). In a few incidences they were found in the diet of leopard in the Cederberg region of South Africa (Martins et al., 2011), but they have no other recorded natural predators in the BMC.

Bat-eared fox are found in grassland habitat with less cover and the majority of their diet consists of insects, followed by plants and fruits, and less often small rodents, birds and reptiles (Kok & Nel, 1992; Stuart et al., 2003). Like Cape fox, bat-eared foxes are persecuted by humans when mistaken for being dangerous and are often captured as by-catch (Nattrass, et al 2017). Bat-eared fox can be vectors of rabies and are preyed upon by many carnivores including leopard, large birds of prey and honey badgers (Begg et al., 2001; Boshoff et al., 1990; Nel, 1993; Smith & Smith, 2015). They are a common road-kill sighting in many parts of South Africa (Bullock et al., 2011).

1.5.4 Herbivores

The Cape grysbok, klipspringer and common duiker have all been described as the dominant ungulate species in the Fynbos biome (Faith, 2011). They are predominantly browsers that select the most nutritious sections of fynbos vegetation (Campbell, 1986; Faith, 2011; Jarman, 1974; Norton, 1980). Duiker have broad diets, browsing either bush or dense grass and sometimes consuming small fauna, while the more nocturnal grysbok are dependent on thick bush (Faith, 2011; Kerley et al., 2010; Stuart & Stuart, 2015). Both can persist near human development and in agricultural lands, where they are pests as they consume new shoots of various crop species (Faith, 2011; Stuart & Stuart, 2015). Palmer et al. (2017) refer to duiker and grysbok as competitors because of their similar feeding methods, habitat requirements and range overlap. Both species can survive in as little as $0.1km^2$ each, with duiker able to survive in a maximum of $1.35km^2$ (Boshoff et al., 2002). Their main predators are leopard and caracal, and their lambs are susceptible to predation by multiple meso-predators (Hayward et al., 2006; Stuart & Stuart 2015).

Although there is some overlap, klipspringer typically inhabit rocky outcrops at a higher altitude than that of duiker or grysbok, therefore competition for food is reduced (Druce et al., 2009; Norton, 1980). Klipspringer are highly selective but nibble on multiple different fynbos species in one browsing session (Jarman, 1974; Norton, 1980; Stuart & Stuart, 2015). Their ranges can be as large as $4.3km^2$ but are often smaller (Boshoff et al., 2002; Norton, 1980). Historically their fur was used for horse saddles which may have pushed population declines, but they are currently of “least concern” on the IUCN red data list (IUCN SSC Antelope Specialist Group, 2016).

Grey rhebok are the largest medium-sized mammal species in the study area, have one of the lowest IUCN red list ratings of “near threatened” and are endemic to South Africa (Birss, 2017; Taylor et al., 2017). This is a newly lowered status, because of their apparent decline in some PAs (Taylor et al., 2017). They represent a potential prey base for local leopard but have been documented in only a few sites in the BMC (Wilkinson, personal communication, 2018). Potential causes of decline include an increase in the number of predators controlling them, such as hunting by domestic dogs and humans (Taylor et al., 2017). Although less frequently found in their diet, leopard do hunt rhebok in the Western Cape (Martins et al., 2011). In the Karoo region, rhebok were found to be important in caracal diets and their lambs have been preyed upon by eagles (Martins et al., 2011; Rautenbach, 2010; Stuart & Stuart, 2015). Rhebok occur in habitats with plateaus and rocky mountain slopes, with grass patches, and at both higher altitudes and in lowland fynbos as they are both browsers and grazers, with preference to recently burnt areas (Apps, 2012; Beukes, 1987; Stuart & Stuart, 2015). Rhebok live in family groups of up to 12 individuals, but individuals each require between 1 and 23.4km^2 of habitat (Boshoff et al., 2002).

Rock hyrax are preyed upon by many predators in the BMC, including leopard, caracal, African wild cat and birds of prey (Avery et al., 2010; Klein & Cruz-Uribe, 1996; Palmer & Fairall, 1988; Wimberger et al., 2009). Hyrax browse and graze on what is available to them, including plants that are low in nutrients and often unpalatable to other herbivores (Shipley, 1999; Stuart & Stuart, 2015). Their middens are where all hyrax in a colony defecate and urinate over generations that fossilise well, providing insight into climatology and the evolutionary history of plants over thousands of years (Carr et al., 2010), and are utilised as herbal remedies (Klein & Cruz-Uribe, 1996). Rock hyrax appear to be abundant in the Fynbos biome, occurring in colonies of up to 60, near sheltered rocks, trees or human structures, along the coast or in high mountains, with small home ranges centred around their shelter (Hoeck et al., 1982; Klein & Cruz-Uribe, 1996; Millar, 1971; Stuart & Stuart, 2015). Rock hyrax can become a pest in agricultural areas and on farms where predators have been removed, as rock hyrax then readily increase (Moran et al., 1987). Humans have been known to hunt them for their meat, hide and other body parts (Klein & Cruz-Uribe, 1996). Disease is another well documented mortality risk to the species (Wimberger et al., 2009), including Sarcoptic mange and dassie bacillus (*Mycobacterium tuberculosis complex sp*.) (Chiweshe, 2005; Hoeck, 1989; Parsons et al., 2008).

Scrub hare *Lepus saxatilis*, Cape hare *Lepus capensis* and Smith's red rock rabbit *Pronolagus rupestris* have been documented as prey species in multiple leopard diet studies in the Western Cape and are also consumed by most carnivores, ranging from the size of African wild cat, and various birds of prey (Chiweshe, 2009; Hayward et al., 2006; Martins et al., 2011; Norton et al., 1986; Smith & Smith, 2015). Scrub hares prefer dense habitats, while still predominantly grazing. Cape hares prefer more open grassland with limited bush present and are primarily grazers but will browse on new plant growth (Stuart & Stuart, 2015). Rock rabbits inhabit rocky outcrops on high altitude, steep mountain sides and generally consume grass (Stuart & Stuart, 2015). Few studies exist with extensive knowledge on this rabbit species. Their home ranges vary between individuals, yet are generally small (Hulbert et al., 1996). Only the Cape hare's density has been estimated at 4 to 24 individuals per 1km^2 in general (Flux & Angermann, 1990).

1.5.5 Introduced mammals

Feral pigs were introduced to pine forests in the BMC in the early 1900's as a failed attempt to control the pine emperor moth *Nudaurelia cytheria cytheria* (Fincham & Hobbs, 2013; Picker & Griffiths, 2011). Since then, the population has increased and is now a class B1 invasive species (Fincham & Hobbs, 2013). However, this species spread is supposedly slower in the BMC than other Mediterranean regions potentially due to predation by leopard (Botha, 1989). Norton (1980) found that the species was often consumed by leopard, contributing to a substantial proportion of their diet in some parts of the BMC. Feral pigs are destructive agricultural pests in that they consume crops and small or juvenile livestock and damage pipes and infrastructure (Fincham & Hobbs, 2013). They have been documented consuming reptiles, birds and small mammal species such as moles and rodents, and their habitat disturbance and competition with insectivores impact native fauna (Fincham & Hobbs, 2013; Hone, 1995; Massei & Genov, 2004).

Feral Cats are a common introduced domestic species that originate from free-roaming pet cats that stray and reproduce (Tennent et al., 2009). Both feral and domestic cats may access natural habitats surrounding agricultural and urban areas. They are generalist feeders, consuming small mammals, birds, amphibians, reptiles and many invertebrates (George, 2010). In the Cape Peninsula, Morling (2014) found small mammals made up the majority of prey species consumed by owned and feral cats. Many larger species do however prey on these cats, including caracal and birds of prey (McPherson et al., 2015; Palmer & Fairall, 1988). Incidence of feral cats preying on any of the medium-sized study species is rare, but competition may exist between meso-predators, and feral/domestic cats may introduce disease to native felids and compromise African wild cats' genetics through possible interbreeding (Erlinge et al., 1984; George, 1974; Le Roux et al., 2015; Morling, 2014; Roelke et al., 1993). Feral cats may be prominent in the buffer zones of the BMC as previous studies have shown that agricultural areas, are where they have high negative impacts on ecosystems (Barratt, 1998).

Feral and free roaming dogs are a medium-sized mammal species that have more recently been reported by agricultural stakeholders and recorded in various nature reserves within the study area (Wilkinson, personal communication, 2018). Little is known about both feral and free-roaming dogs in the Fynbos biome, however globally feral dogs are a well-documented problem, that introduce various threats into natural ecosystems and to various mammal species occurring in these systems (Young et al., 2011). These include competition for food and habitat resources with natural carnivores, predation of mammals and other fauna, and disease transmission (Butler et al., 2003). These threats are potentially difficult to mitigate/eradicate once established (Butler et al., 2003; Sakai et al., 2001; Young, et al., 2011). With the current low density of mammalian species and the many threats which already exist for these species, it is easy for these dogs (especially when exhibiting pack-behaviour) to outcompete leopard for vital resources and cause ecological damage on multiple levels (Less et al., 2016).

1.6 Study Importance

There is a deficiency of knowledge on many scales and levels pertaining to faunal ecology in the Fynbos biome. The majority of the 29 medium-sized, mammalian study species are poorly understood in general and many of their ecologies have not been assessed in-depth within this fynbos ecosystem. Previous studies in the BMC have focused on either an aspect of natural ecology or human developments, yet few studies consider/examine the effects of human development on faunal ecology, within the buffer zones (as opposed to the core zones). This study is a necessary broad assessment to detect what future resources, research and management planning need to be allocated and how. It analyses the historic, present and future dynamics in a mosaic landscape of fynbos habitats and various human land-covers, while highlighting acute threats, impacts and solutions. By utilising stakeholders as information sources of the study, it allows all knowledge attained to be more accessible to the people directly exposed to the ecosystems that they live within.

The few ecological studies which focus on local leopard populations present grounds to base and build this study on. These studies highlighted that leopard habitats and prey loss are concerns that need further analysis (Martins & Martins, 2006; Nieman, 2018). Considering the BMC's growing expanses of development and human population size, it thus motivates that a strong and in-depth understanding is required of current medium-sized mammal ecosystems and how they are altered by anthropogenic activities, in order to mitigate future unnatural losses to mammal populations. The inclusion and considerations of the two UNESCO Biosphere Reserves, the BMC Heritage Site and many landowners allows the study to manage conservation threats, implement strategic plans and research requirements and enforce these plans and research through potentially ideal mediums (Batisse, 1982).

1.7 Primary goals, aims, objectives and hypothesizes

The primary goal of this study was to understand whether human development threats to the local leopard population in the BMC's buffer habitats, are of significant concern to the survival of the species. Specific threats include: the alteration of the leopards' prey-base presence, effects on distribution, abundance, and diversity and threats of habitat loss to either leopard and/or their prey. This study will provide valuable data to be incorporated into a management plan for the leopard population and other medium-sized mammals in the BMC. This study aimed to assess a mammalian landscape with a multi-scale and multi-species approach to improve future conservation planning.

A: How have land-cover patterns changed from 1990 to 2013, in relation to habitats utilised by leopards. Primary objectives under this aim were to:

1.) Determine how land-cover distributions have changed with time in the BMC and how this may affect the natural vegetation class (available habitats).

2.) Identify areas which have undergone significant development and what key habitats have been altered and/or may be at risk of further encroachment.

3.) Determine if there are significant differences in the proportions and sizes of areas where land-use has changed: between the two biosphere reserves, the seven MZ's and at varying distances from landmarks (roads, settlement edges, protected boundaries) in the BMC

B: Asses spatial changes in the frequency of incidental wildfire observations from 1957 to 2017, including patterns and veld ages in potential leopard habitat, with the objectives to:

1.) Establish if and how wildfire frequencies are changing in the BMC.

2.) Determine if there are any differences in the frequency and locality of fires.

3.) Assess total areas of habitat that have been burnt at a 15-year period, to determine if there have been changes in habitat availability for mammals.

C: Establish a baseline understanding of medium-sized mammal composition, distributions and population changes over time as perceived by stakeholders in the buffer zones, with objectives to:

1.) Document stakeholder sightings of species distributions and whether there were differences in the occurrence of any species across the landscape.

2.) Highlight what changes, threats and impacts to medium-sized mammals were observed from stakeholder responses on private properties, to assess if and which mammal species may be at risk of a population decline and what further resources and research are required.

3.) Determine what landscape factors were influencing the perceived presence of and changes to medium-sized mammal populations.

1.8 Reference list

Admasu, E., Thirgood, S.J., Bekele, A. & Laurenson, M.K. (2004). A note on the spatial ecology of African civet *Civettictis civetta* and common genet *Genetta genetta* in farmland in the Ethiopian Highlands. *African Journal of Ecology*, 42(2), 160-162.

Alphan, H. (2017) Analysis of road development and associated agricultural land change. *Environmental Monitoring and Assessment*, 190(5), 1-11.

Anderson, P.M. & O'Farrell, P.J. (2012). An ecological view if the history of the City of Cape Town. *Ecology and Society*, 17(3), 28.

Apps, P. (2012). *Smither's mammals of Southern Africa field guide*. Cape Town: Struik Nature.

Archibald, S., Staver, A.C. & Levin, S.A. (2012). Evolution of human-driven fire regimes in Africa. *Proceedings of National Academy of Sciences*, 109(3), 847-852.

Athreya, V., Odden, M., Linnell, J.D.C., Krishnaswamy, J. & Karanth, U. (2013). Big cats in our backyards: Persistence of large carnivores in a human dominated landscape in India. *PLoS ONE*, 8(3), 1-8.

Auld, T.D. & Denham, A.J. (2001). The impact of seed predation by mammals on post-fire seed accumulation in the endangered shrub *Grevillea caleyi (Proteaceae*). *Biological Conservation*, 97(3), 377-385.

Avenant, N.L. & Nel, J.A.J. (1997). Prey use by four syntopic carnivores in a strandveld ecosystem. South *African Journal of Wildlife Research*, 27(3), 86-93.

Avery, G. Robertson, A.S., Palmer, N.G. & Prins, A.J. (2010). Prey of giant eagle owls in the De Hoop Nature Reserve, Cape Province, and some observations on hunting strategy. *Ostrich*, 56(1-3), 117-122.

Baily, T.N. (1993). *The African leopard: Ecology and behavior of a solitary felid*. New York: Columbia University Press.

Baker, C.M. (1997). Communication in marsh mongooses (*Atilax paludinosus*): Anal gland secretion and scat discrimination in adults, and individual variation in vocalizations of juveniles. *South African Journal of Zoology*, 33(1), 49-51.

Balme, G., Hunter, L. & Slotow, R. (2007). Feeding habitat selection by hunting leopards Panthera pardus in a woodland savanna: Prey catchability versus abundance. *Animal Behaviour*, 74(3), 589-598.

Balme, G.A., Lindsey, P.A., Swanepoel, L.H. & Hunter, L.T.B. (2014). Failure of research to address the range wide conservation needs of large carnivores: Leopards in South Africa as a case study. *Conservation Letters*, 7(1), 3-14.

Bands, D.P. (1977). Prescribed burning in Cape fynbos catchments. In H.A. Mooney & C.E. Conrad (Eds.), *Symposium on the environmental consequences of fire and fuel management in Mediterranean ecosystems* (pp. 245-256). Washington, D.C.: USDA Forest Service,

Barnard, A. (1992). *Hunters and herders of Southern Africa: A comparative ethnography of the Khoisan peoples*. Melbourne: Cambridge University Press.

Barratt, D. G. (1998). Predation by house cats, *Felis catus (L.),* in Canberra, Australia. II. Factors affecting the amount of prey caught and estimates of the impact on wildlife. *Wildlife Research*, 25, 475.

Barro, S.C. & Conard, S.G. (1991). Fire effects on California chaparral systems: An overview. *Environment International*, 17, 135–149.

Barthelmess, E.L. (2006). Hystrix africaeaustralis. *Mammalian Species*, 788, 1-7.

Batisse, M. (1982). The biosphere reserve: A tool for environmental conservation management. *Environmental Conservation*, 9(2), 101-111.

Begg, C.M., Begg, K.S., Du Toit, J.T. & Mills, M.G.L. (2001). Sexual and seasonal variation in the diet and foraging behavior of a sexually dimorphic carnivore, the honey badger (*Mellivora capensis*). *Journal of Zoology, London*, 260, 301-316.

Bender, D.J., Contreras, T.A. & Fahrig, L. (1998). Habitat loss and population decline: A meta-analysis of the patch size effect. *Ecology*, 79(2), 517-33.

Beukes, P.C. (1987). Responses of grey rhebok and bontebok to controlled fires in coastal renosterveld. *South African Journal of Wildlife Research*, 17(3), 103-108.

Bigalke, R.C. & Willan, K. (1984). Effects of fire regime on faunal composition and dynamics. In P.V. de Booysen & N.M. Tainton (Eds.), *Ecological Effects of Fire in South African Ecosystems* (pp. 255-271). Berlin: Springer.

Biosphere reserves - myth or reality? (1996). Unpublished paper delivered at the Proceedings of the Workshop on Biosphere Reserves, World Conservation Congress. Montreal, October 13-23.

Birss, C. (2017). Mammals. In: Turner, A.A. (Ed.), *Western Cape Province State of Biodiversity 2017* (pp. 191-230). Stellenbosch: CapeNature Scientific Services.

Bond, W.J. & van Wilgen, B.W. (1996). *Fire and Plants*. London: Chapman and Hall.

Bond, W.J., Ferguson, M. & Forsyth, G. (1980). Small mammals and habitat structure along altitudinal gradients in the southern Cape mountains. *South African Journal of Zoology*, 15(1), 33-43.

Bond, W.J., Woodward, F.I. & Midgley, G.F. (2005). The global distribution of ecosystems in a world without fire. *New Phytologist*, 165, 525-537.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2001). A pragmatic approach to estimating the distributions and spatial requirements of the medium- to large-sized mammals in the Cape Floristic Region, South Africa. *Diversity and Distributions*, 7, 29-43.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2002). Estimated spatial requirements of the medium- to large-sized mammals, according to broad habitat units, in the Cape Floristic Region, South Africa. *African Journal of Range and Forage Science*, 19(1), 29-44.

Boshoff, A.F., Palmer, N.G. & Avery, G. (1990). Variation in the diet of martial eagles in the Cape Province, South Africa. *South African Journal of Wildlife Research*, 20, 57-68.

Botha, S.A. (1989). Feral pigs in the Western Cape Province: Failure of a potentially invasive species. *South African Forestry Journal*, 151(1), 17-25.

Bothma, J. du P. (2012). A review of the ecology of the southern Kalahari leopard. *Transactions of the Royal Society of South Africa*, 53 (2), 267-266.

Bothma, J. du P., van Rooyen, N. & Le Riche, E.A.N. (1997). Multivariate analysis of the hunting tactics of Kalahari leopards. *Koedoe*, 40 (1), 41-56.

Boucher, C. & Moll, E. (1980). South African Mediterranean shrublands. In F. Di Castri, D.W. Goodall & R.L. Specht (Eds.), *Ecosystems of the World II: Mediterranean-type shrublands* (pp. 233-246). Amsterdam: Elsevier Scientific Publishing Company.

Braczkowski, A., Watsom, L., Coulson, D., Lucas, J., Peiser, B. & Rossi, M. (2012a). The diet of caracal, Caracal caracal, in two areas of the southern Cape, South Africa as determined by scat analysis. *South African Journal of Wildlife Research*, 42(2), 111-116.

Braczkowski, A., Watsom, L, Coulson, D. & Randall, R. (2012b). Diet of leopards in the southern Cape, South Africa. *African Journal of Ecology*, 50, 377-380.

Breytenbach, G.J. & Skinner, J.D. (1982). Diet, feeding and habitat utilization by bushpigs *Potamochoerus porcus* Linnaeus. *South African Journal of Wildlife Research*, 12, 1-7.

Bronikowski, A.M. & Altmann, J. (1996). Foraging in a variable environment: Weather patterns and the behavioral ecology of baboons. *Behavioral Ecology and Sociobiology*, 39(1), 11-25.

Brooks, T.M., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A.B., Rylands, A.B., Konstant, W.R. et al. (2002). Habitat loss and extinction hotspots of biodiversity. *Conservation Biology*, 16(4), 909-923.

Bullock, K.L., Malan, G. & Pretorius, M.D. (2011). Mammal and bird road mortalities on the Upington to Twee Rivieren main road in the southern Kalahari, *South Africa. African Zoology*, 46(1), 60-71.

Buskirk, S.W. (1999). Mesocarnivores of Yellowstone. In T.W. Clark, P.M. Curlee, S.C. Minta & P.M. Kareiva, (Eds.) *Carnivores in Ecosystems: The Yellowstone Experience (*pp. 165-188). Connecticut: Yale University Press.

Busse, C. (1980). Leopard and lion predation upon chacma baboons living in the Moremi Wildlife Reserve. *Botswana Notes & Records*, 12.

Butler, J.R.A., du Toit, J.T. & Bingham, J. (2003). Free-ranging domestic dogs (Canis familiaris) as predators and prey in rural Zimbabwe: Threats of competition to large wild carnivores. *Biological Conservation*, 115, 369-378.

Campbell, B.M. (1986). Plant spinescence and herbivory in a nutrient poor ecosystem. *Oikos*, 47(2), 168-172.

Cardillo, I., Mace, G.M., Jones, K.E., Bielby, J., Bininda-Emonds, O.R.P., Sechrest, W. et al. (2005). Multiple causes of high extinction risk in large mammal species. *Science*, 309, 1239-1241.

Caro, T.M. (2003). Umbrella species: Critique and lessons from East Africa. *Animal Conservation*, 6, 171-181.

Caro, T.M. & Stoner, C.J. (2004). The potential for interspecific competition among African carnivores. *Biological Conservation*, 110, 67-75.

Cavallini, P. & Nel, J.A.J. (1990). The feeding ecology of the Cape grey mongoose, *Galerella pulverulenta* (Wagner 1839) in a coastal area. *African Journal of Ecology*, 28, 123-130.

Cavallini, P. & Nel, J.A.J. (1995). Comparative behaviour and ecology of two sympatric mongoose species (Cynictis penicillata and Galerella pulverulenta). *Journal of Zoology*, 30 (2), 46-49.

Chan, C., van Vuuren, B.J. & Cherry, M.I. (2011). Fynbos fires may contribute to the maintenance of high genetic diversity in orange breasted sun birds (*Anthobaphes violacea*). *South African Journal of Wildlife Research*, 41(1), 87-94.

Chevallier, N. & Ashton, B. (2006). *A report on porcupine quill trade in South Africa*. London: International fund for Animal Welfare.

Chiweshe, N. (2005). An update on the annual dassie census in Matobo Hills, Zimbabwe. *Afrotherian Conservation*, 3, 7–8.

Chiweshe, N. (2009). Black Eagles and hyraxes — the two flagship species in the conservation of wildlife in the Matobo Hills, Zimbabwe. *Journal of African Ornithology*, 78(2), 381-386.

Conradie, B. & Piesse, J. (2013). The effects of predator culling on livestock losses: Ceres, South Africa, 1979-1987. *African Journal of Agricultural and Resource Economics*, 8, 265-274.

Conradie, D.C.U. (2012). *South Africa's climatic zones: Today, tomorrow*. Unpublished paper delivered at International Green Building Conference and Exhibition on Future Trends and Issues Impacting on the Built Environment. Sandton, July 25-26.

Cooper, R.L. & Skinner, J.D. (1979). Importance of Termites in the diet of the Aardwolf *Proteles Cristatus* in South Africa. *South African Journal of Zoology*, 14(1), 5-8.

Cowling, R.M. (1992). *The ecology of fynbos: Nutrients, fire and diversity*. Cape Town: Oxford University Press.

Cowling, R.M., Macdonald, I.A.W. & Simmons, M.T. (1996). The Cape Peninsula, South Africa: Physiographical, biological and historical background to an extraordinary hot-spot of biodiversity. *Biodiversity and Conservation*, 5, 527-550.

Cowlishaw, G. (1994). Vulnerability to predation in baboon populations. *Behaviour*, 131(3/4), 194-304.

Creel, S. (2001). Four factors modifying the effect of competition on carnivore population dynamics as illustrated by African wild dogs. *Conservation Biology,* 15, 271–274.

DeVault, T.L., Olson, Z.H., Beasley, J.C. & Rhodes Jr., O.E. (2011). Mesopredators dominate competition for carrion in an agricultural landscape. *Basic and Applied Ecology*, 12, 268-274.

Do Linh San, E., Begg, C., Begg, K. & Abramov, A.V. (2016*). Honey badger. Mellivora capensis* [Online]. Retrieved April 29, 2019: https://www.iucnredlist.org/species/41629/45210107.

Donaldson, R., van Niekerk, A., du Plessis, D. & Spocter, M. (2012). Non-metropolitan growth potential of Western Cape municipalities*.* *Urban Forum*, 23, 367-389.

Druce, D.J., Brown, J.S., Kerley, G.I.H, Kotler, B.P., Mackay, R.L. & Slotow, R. (2009). Spatial and temporal scaling in habitat utilization by klipspringers (*Oreotragus oreotragus*) determined using giving-up densities. *Austral Ecology*, 34, 577-587.

Duffy, E., Cardinale, B.J., France, K.E., McIntyre, P.B., Thébault & Loreau, M. (2007). The functional role of biodiversity in ecosystems: Incorporating trophic complexity. *Ecology Letters*, 10, 522-538.

Eby, S.L., Anderson, T.M., Mayember, E.P. & Ritchie, M.E. (2014). The effect of fire on habitat selection of mammalian herbivores: The role of body size and vegetation characteristics. *Journal of Applied Ecology*, 83, 1196-1205.

Eigenbrod, F, Hecnar, S.J. & Fahrig, L. (2008). Accessible habitat: An improved measure of the effects of habitat loss and roads on wildlife populations. *Landscape Ecology*, 23, 159-168.

Elmhagen, B., Ludwig, G., Rushton, S.P., Helle, P. & Lindén, H. (2010). Top predators, mesopredators and their prey: Interference ecosystems along bioclimatic productivity gradients. *Journal of Animal Ecology*, 79, 785-794.

Elias, S.A. (2014). Rise of human influence on the world's biota. *Reference Module in Earth Systems and Environmental Science.* Oxford: Elsevier

Erlinge, W., Göransson, G., Högstedt, G., Jansson, G., Liberg, O., Loman, G. et al. (1984). Can vertebrate predators regulate their prey? *American Naturalist*, 123, 125-133.

Estes, J.A., Terborgh, J., Brashares, J.S., Power, M.E., Berger, J., Bond, W.J. et al. (2011). Trophic downgrading of planet Earth. *Science*, 333, 301–306

Fairbanks, D.H.K., Hughes, C.J. & Turpie, J.K. (2004). Potential impact of viticulture expansion on habitat types in the Cape Floristic Region, South Africa. *Biodiversity and Conservation*, 13, 1075-1100.

Faith, J.T. (2011). Ungulate community richness, grazer extinctions, and human subsistence behaviour in southern Africa's Cape Floral Region. *Palaeogeography, Palaeoclimatology, Palaeocology*, 306, 219-227.

Fehlmann, G., O'Riain, M.J., Kerr-Smith, C. & King, A.J. (2017). Adaptive space use by baboons (*Papio ursinus*) in response to management interventions in a human-changed landscape. *Animal Conservation*, 20, 101-109.

Fincham, J.E. & Hobbs, J.A. (2013). Feral pigs: A threat to ground nesting birds. *Ornithological Observations*, 4, 83-89.

Flux, J.E.C. & Angermann, R. (1990). The hares and jackrabbits. In J.A. Chapman & J.E.C. Flux (Eds.), *Rabbits, hares and pikas. Status survey and conservation action plan* (pp. 61-94). Gland: International Union for Conservation of Nature and Natural Resources.

Folke, C., Hahn, T., Olsson, P & Norberg, J. (2005). Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources*, 30, 441-473.

Forsyth, G.G. & van Wilgen, B. (2008). The recent fire history of Table Mountain National Park and implication for fire management. *Koedoe*, 50, 3-9.

Fraser, M.W. (1990) Small mammals, birds and ants as seed predators in post-fire mountain fynbos. *South African Journal of Wildlife Research*, 20(2), 52-56.

Frohlich, M. (2011) *Studying the foraging ecology of leopards (Panthera pardus) using activity and location data: An exploratory attempt.* Unpublished master's thesis, Humboldt University of Berlin.

Gaubert, P., Carvalho, F., Camps, D. & Do Linh San, E. (2015). *Common genet Genetta genetta* [Online]. Retrieved April 29, 2019: https://www.iucnredlist.org/species/41698/45218636

Gavashelishvili, A. & Lukarevskiy, V. (2008). Modeling the habitat requirements of leopard *Panthera pardus* in west and central Asia. *Journal of Applied Ecology*, 45, 579-588.

George, S. (2010). *Cape Town's domestic cats: Prey and movement patterns in deep-urban and urban edge areas*. Unpublished master's thesis, University of Cape Town.

George, W. G. (1974). Domestic cats as predators and factors in winter shortages of raptor prey. *The Wilson Bulletin*, 86, 384–396.

Geyer, H., Schloms, B., du Plessis, D. & van Eeden, A. (2011). Land quality, urban development and urban agriculture within the Cape Town urban edge. *Town and Regional Planning,* 59, 43–55.

Green, D.S. (2015). *Aardwolf Proteles* cristata [Online]. Retrieved March 17, 2019: https://www.iucnredlist.org/species/18372/45195681

Grobler, J.H. (1981). Feeding behaviour of the caracal *Felis caracal* Schreber 1776 in the Mountain Zebra National Park. *South African Journal of Zoology*, 16(4), 259-262.

Gumbi, D.P. (2011). *The impact of change in climate, human demography, and other social factors on the fire regime of the Kogelberg Nature Reserve*. Unpublished doctoral dissertation, University of Kwazulu-Natal, South Africa.

Halpern, A.B.W. & Meadows, M.E. (2013). Fifty years of land use change in the Swartland, Western Cape, South Africa: Characteristics, causes and consequences. *South African Geographical Journal*, 95(1), 38-49.

Hannah, L., Roehrdanz, P.R., Ikegami, M., Shepard, A.V., Shaw, M.R., Tabor, G. et al. (2013). Climate change, wine and conservation. *Proceedings of National Academy of Sciences*, 110(17), 6907-6912.

Hansen, A.J. & DeFries, R (2007). Ecological mechanisms linking protected areas to surrounding lands. *Ecological Applications*, 17(4), 974-988.

Happold, D.C.D. (2013). Hystrix africaeaustralis. Cape crested porcupine (Cape porcupine). In D.C.D. Happold (Ed.), *Mammals of Africa. Volume III: Rodents, hares and rabbits* (pp. 676-678). London: Bloomsbury Publishing.

Harrison, S. & Bruna, E. (1999). Habitat fragmentation and large-scale conservation: What do we know for sure? *Ecography*, 22, 225-232.

Hayward, M.W., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G. & Kerley, G.I.H. (2006). Prey preferences of the leopard (*Panthera pardus*). *Journal of Zoology*, 270, 298-313.

Heijnis, C.E., Lombard, A.T., Cowling, R.M. & Desmet, P.G. (1999). Picking up the pieces: A biosphere reserve framework for a fragmented landscape - The Coastal Lowlands of the Western Cape, South Africa. *Biodiversity and Conservation*, 8, 471-496.

Herbst, M. & Mills, M.G.L. (2010). The feeding habits of the African wildcat (*Felis silvestris cafra*), a facultative trophic specialist, in the southern Kalahari (Kgalagadi Transfrontier Park, South Africa/Botswana). *Journal of Zoology*, 280, 403-413.

Hoeck, H.N. (1989). Demography and competition in hyrax. *Oecologia*, 79, 353–360.

Hoeck, H.N., Klein, H. & Hoeck, P. (1982). Flexible social organization in hyrax. *Journal of Animal Psychology*, 59, 265-298.

Hoffman, T.S. & O' Riain, M.J. (2012). Troop size and human-modified habitat affect the ranging patterns of a chacma baboon population in the Cape Peninsula, South Africa. *American Journal of Primatology*, 74, 853- 863.

Hone, J. (1995). Spatial and temporal aspects of vertebrate pest damage with emphasis on feral pigs. *Journal of Applied Ecology*, 32, 311-319.

Hulbert, I. A. R., Iason, G. R., Elston, D. A. & Racey, P. A. (1996). Home-range sizes in a stratified upland landscape of two lagomorphs with different feeding strategies. *Journal of Applied Ecology*, 33, 1479-1488.

Hunter, L., Henschel, P. & Ray, J.C. (2013). *Panthera pardus* leopard. In J. Kingdon & M. Hoffmann (Eds.), *Mammals of Africa. Volume V: Carnivores, pangolins, equids and rhinoceroses* (pp. 159-168). London: Bloomsbury Publishing.

Ishwaran, N., Persic, A. & Tri, N.H. (2008). Concept and practice: A case of UNESCO Biosphere Reserves. *International Journal of Environment and Sustainable Development*, 7(2), 118-131.

Jacobson, A.P., Gerngross, P., Lemeris Jr., J.R., Schoonover, R.F. Anco, C., Breitenmoser-Wursten, C. et al. (2016). Leopard (*Panthera paradus*) status, distribution and research efforts across its range. *PeerJ*, 1974, 1-28.

Jacques, H., Reed-Smith, J. & Somers, M.J. (2015). *Aonyx capensis African clawless otter*. [Online]. Retrieved July 12, 2019: https://www.iucnredlist.org/species/1793/21938767

Jaffe, K.E. & Isabell, L.A. (2009). After the fire: Benefits of reduced ground cover for vervet monkeys (*Cercopithecus aethiops*). *American Journal of Primatology*, 71, 252-260.

Jarman, P.J. (1974). The social organisation of antelope in relation to their ecology. *Behaviour,* 48(4), 215-267.

Johnson, M.R., Anhaeusser, C.R. & Thomas, R.J. (2006). *The geology of South Africa*. Johannesburg: Geological Society of South Africa.

Kamler, J.F. & Macdonald, D.W. (2014). Social organization, survival, and dispersal of cape foxes (*Vulpes chama*) in South Africa. *Mammalian Biology*, 79, 64-70.

Kamler, J.F., Stenkewitz, U. & Macdonald, D.W. (2013). Lethal and sublethal effects of black-backed jackals on cape foxes and bat-eared foxes. *Journal of Mammalogy*, 94(2), 295-306.

Kamler, J.F., Stenkewitz, U., Klare, U., Jacobsen, N.F. & Macdonald D.W. (2012). Resource Partitioning among Cape foxes, bat-eared foxes, and black-backed jackals in South Africa. *The Journal of Wildlife Management*, 76(6), 1241-1253.

Keeley, J.E., Pausas, J.G., Rundel, P.W., Bond, W.J. & Bradstock, R.A. (2011). Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science*, 16(8), 1360-1385.

Kerley, G.I.H., Landman, M. & de Beer, S. (2010). How do small browsers respond to resource changes? Dietary response of the Cape grysbok to clearing alien *Acacias*. *Functional Ecology*, 24, 670-275.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. (2003). Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112, 169-190.

Khan, U., Lovari, S., Ali Shah, S. & Ferretti, S. (2018). Predator, prey and humans in a mountainous area: Loss of biological diversity leads to trouble. *Biodiversity and Conservation*, 27(11), 2795-2813.

Kintz, D.B., Young, K.R. & Crews-Meyer, K.A. (2006). Implications of land use/land cover change in the buffer zone of a national park in tropical Andes. *Environmental Management*, 38(2), 238-252.

Klein, R.G. & Cruz-Uribe, K. (1996). Size variation in the rock hyrax (*Procavia capensis*) and late quaternary climatic change in South Africa. *Quaternary Research*, 46, 193-207.

Klein, R.G. (1983). Palaeoenvironmental implications of quaternary large mammals in the fynbos region. *South African National Scientific Programmes Reports*, 75, 116-138.

Kok, O.B. & Nel, J.A.J. (1992). Diet of bat-eared fox in the Orange Free State and Northern Cape Province. *South African Journal of Natural Science and Technology*, 22(2), 36-39.

Kraaij, T., Cowling, R.M. & van Wilgen, B.W. (2011). Paste approaches and future challenges to the management of fire and invasive alien plants in the new Garden Route National Park. *South African Journal of Science*, 107(9/10), 1-11.

Kruuk, H. & Mills, M.G.L. (1983). Notes on food and foraging of the honey badger *Mellivora capensis* in the Kalahari Gemsbok National Park. *Koedoe*, 26, 153-157.

Kruuk, H. & Moorhouse, A. (1991). The spatial organization of otters (*Lutra lutra*) in Shetland. *Journal of Zoology, London*, 224, 41–57.

Lambert, T.D., Malcolm, J.R. & Zimmerman, B.L. (2006). Amazonian small mammal abundances in relation to habitat structure and resource abundance. *Journal of Mammalogy*, 87(4), 667-776.

Larivière, S. (2001). Mammalian species: *Aonyx capensis*. *The American Society of Mammalogists*, 671, 1-6.

Le Roux, J.J., Foxcroft, L.C., Herbst, M. & MacFadyen, S. (2015). Genetic analysis shows low levels of hybridization between African wildcats (*Felis silvestris lybica*) and domestic cats (*F. s. catus*) in South Africa*.* *Ecology and Evolution*, 5(2), 288-299.

Legge, S., Murphy, S., Heathcote, J., Flaxman, E., Augustyn, J. & Crossman, M. (2008). The short-term effects of an extensive and high-intensity fire on vertebrates in the tropical savannas of the central Kimberley, northern Australia. *Wildlife Research*, 35, 33-43.

Letni, M., Koch, F., Gordan, C., Crowther, M.S. & Dickman, C.R. (2009). Keystone effects of an alien top-predator stem extinctions of native mammals. *Proceedings of the Royale Society B*, 276, 3249-3256.

Li, W., Wang, Z. & Tang, H. (1999). Designing the buffer zone of a nature reserve: A case study in Yancheng Biosphere Reserve, China. *Biological Conservation*, 90, 159-165.

Lindsay, P. (2008). *A spatio-temporal analysis of the habitat use of leopards (Panthera pardus) in the Karoo biome of the Cederberg Mountains, South Africa.* Unpublished honours' thesis, University of Cape Town.

Lloyd, P.A. & Millar, J.C.G. (1981). A questionnaire survey of some of the larger mammals of the Cape Province. *Bontebok*, 1, 1-58.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Agulhas Plain, South Africa. *Conservation Biology*, 11(5), 1101-1116.

Lovari, S., Boesi, R., Minder, I., Mucci, N., Randi, E. Dematteis, A. et al. (2009). Restoring a keystone predator may endanger a prey species in a human-altered ecosystem: The return of the snow leopard to Sagarmatha National Park. Animal Conservation, 12(6), 559-570.

Lovell, S.T. & Johnston, D.M. (2009). Designing landscapes for performance based on emerging principles in landscape ecology. *Ecology and Society*, 14(1), 44.

Macdonald, D.W., Loveridge, A.J. & Nowell, K. (2010). Dramatis personae: An introduction to the wild felids. In D.W. Macdonald & A.J. Loveridge (Eds.), *Biology and Conservation of Wild Felids* (pp. 3-58). New York: Oxford University Press.

Mack, M.C. & D'Antonia, C.M. (1998). Impacts of biological invasions on disturbance regimes. *Trends in Ecology and Evolution*, 13, 195-198.

Makenzi, P.M. (2013). The biosphere reserve concept as a tool for sustainable natural resource management in the Eastern African region. In R. Pool-Stanvliet & M. Clüsener-Godt (Eds.), *AfriMAB: Biosphere Reserves in Sub-Saharan Africa: Showcasing sustainable development* (pp. 1-14). Pretoria: Department of Environmental Affairs & Paris: UNESCO.

Mangnall, M.J. & Crowe, T.M. (2003). The effects of agriculture on farmland bird assemblages on the Agulhas Plain, Western Cape, South Africa. *African Journal of Ecology*, 41, 266-276.

Mann, G.K.H., Wilkinson, A., Hayward, J., Drouilly, M., O'Riain, M.J. & Parker, D.M. (2019). The effects of aridity on land use, biodiversity and dietary breadth in leopards. *Mammalian Biology*, 98, 43-51.

Marker, L.L. & Dickman, A.J. (2005). Factors affecting leopard (*Panthera pardus*) spatial ecology, with particular reference to Namibian farmlands. *South African Journal of Wildlife Research*, 35(2), 105-115.

Martins, Q. & Harris, S. (2013). Movement, activity and hunting behavior of leopards in the Cederberg mountains, South Africa. *African Journal of Ecology*, 51, 571-579.

Martins, Q. & Martins, N. (2006). Leopards of the Cape: Conservation and conservation concerns. *International Journal of Environmental Studies*, 63(5), 579-585.

Martins, Q. (2010). *The ecology of the leopard Panthera pardus in the Cederberg Mountains.* Unpublished doctorate dissertation, University of Bristol.

Martins, Q., Horsnell, W.G.C., Titus, W., Rautenbach, T. & Harris, S. (2011). Diet determination of the Cape Mountain leopards using global positioning system location clusters and scat analysis. *Journal of Zoology*, 283, 81-87.

Massei, G. & Genov, P.V. (2004). The environmental impact of wild boar. *Galemys*, 16, 135-145.

Matthews, T. (2002). South African micromammals and predators: Some comparative results. *Archaeometry*, 44(3), 364-370.

McPherson, S.C., Brown, M & Downs, C.T. (2015). Diet of the crowned eagle (*Stephanoaetus coronatus*) in an urban landscape: Potential for human-wildlife conflict. *Urban Ecosystem*, 19, 383-396.

McRae, B.H., Dickson, B.G., Keitt, T.H. & Shah, V.B. (2008) Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecological Society of America, 89, 2712–2724.

Melville, H.I.A.S., Bothma, J. du P. & Mills, M.G.L. (2004). Prey selection by caracal in the Kgalagadi Transfrontier Park. *South African Journal of Wildlife Research*, 34(1), 67-75.

Millar, R.P. (1971). Reproduction in the rock hyrax (*Procavia capensis*). *Zoologica Africana*, 6(2), 243-261.

Mills, M.G.L. (1984). Prey selection and feeding habits of the large carnivores in the southern Kalahari. *Koedoe*, 281-294.

Minnich, R.A. & Chou, Y.H. (1997). Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. *International Journal of Wildland Fire*, 7, 221-248.

Mitchell, M. (2014). Chapter 3: The first roads- building the foundation for a country-wide road network. In M. Mitchell (Ed.). *A brief history of transport infrastructure in South Africa up to the end of the 20th century* (pp. 41-45). Midrand: South African Institution of Civil Engineering.

Mooney, H.A., Lubchenco, J., Dirzo, R. & Sala, O.E. (1995). Biodiversity and ecosystem functioning: Basic principles. In: Heywood, V.H. (Ed.), *Global biodiversity assessment (pp*.281-325). Cambridge: Cambridge University Press.

Moran, S., Sofer, S. & Cohen, M. (1987). Control of the rock hyrax, *Procavia capensis*, in fruit orchards by fluoroacetamide baits. *Crop Protection*, 6(4), 265-270.

Morling, F. (2014). *Cape Town's cats: reassessing predation through kitty-cams*. Unpublished master's thesis, University of Cape Town.

Morris, D.W. (2005). Paradoxical avoidance of enriched habitats: Have we failed to appreciate omnivores? *Ecology*, 86(10), 2568-2577.

Mucina, L., Rutherford, M.C. & Powrie, L.W. (2005). *Vegetation map of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.

Mukherjee, S., Goyal, S.P., Johnsingh, A.J.T. & Pitman, M.R.P.L. (2004). The importance of rodents in the diet of jungle cat (*Felis chaus*), caracal (*Caracal caracal*) and golden jackal (*Canis aureus*) in Sariska Tiger Reserve, Rajasthan, India. *Journal of Zoology, London*, 262, 405-411.

Muller, K. (2008). Assessing cooperative environmental governance systems: The cases of Kogelberg Biosphere Reserve and the Olifants-Doorn Catchement Management Agency. *Politea*, 27(1), 86-104.

Murray, T. (2015). *Mega structures and master minds: Great feats of civil engineering in Southern Africa*. Cape Town: Tafelberg.

Musakwa, W. & Wang, S. (2018). Landscape change and its drivers: A Southern African perspective. *Current Opinion in Environmental Sustainability*, 33, 80-86.

Nattrass, N., Conradie, B., Drouilly, M. & O'Riain, M.J. (2017). A brief history of predators, sheep farmers and government in the Western Cape, South Africa. Centre for Social Research, University of Cape Town.

Nel, J.A.J. (1984). Behavioural ecology of canids in south western Kalahari. *Koedoe*, 229-235.

Nel, J.A.J. (1993). The bat-eared fox: A prime candidate for rabies vector? *Onderstepoort Journal of Veterinary Research*, 60, 395-397.

Nguyen, N., Bosch, O. & Maani, K. (2009*). The importance of systems thinking and practice for creating biosphere reserves as "learning laboratories for sustainable development."* Unpublished paper delivered at the Fifty-Third Proceedings of the International Society for the Systems Sciences Conference. Queensland, July 1-17.

Nieman, W.A. (2018). *Culture, conflict and cuisine: A quantitative assessment of terrestrial vertebrate off-take at the human-wildlife interface.* Unpublished master's thesis, Stellenbosch University.

Nieman, W.A., Leslie, A.J., Wilkinson, A. & Wossler, T.C. (2019). Socioeconomic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa. *Journal of Nature Conservation*, 52, 1-7.

Norton, P.M. & Lawson, A.B. (1985). Radio-tracking of leopards and caracals in the Stellenbosch area, Cape Province. *South African Journal of Wildlife Research*, 15(1), 17-24.

Norton, P.M. (1980). *The habitat and feeding ecology of the klipspringer Oreotragus oreotragus (Zimmerman, 1780) in two areas of the Cape Province.* Unpublished master's thesis, University of Pretoria.

Norton, P.M., Lawson, A.B., Henley, S.R. & Avery G. (1986). Prey of leopards in four mountainous areas of the south-western Cape Province. *South African Journal of Science and Technology*, 16(2), 47-52.

Noss R.F. (1990). Indicators for monitoring biodiversity; a hierarchical approach. *Conservation Biology*, 12, 355–364.

Novellie, P.A., Manson, J. & Bigalke, R.C. (1983). Behavioural ecology and communication in the Cape grysbok. *South African Journal of Zoology*, 19(1), 22-30.

O'Laughlin, B., Bernstein, H., Cousins, B. & Peteres, P.E. (2013). Introduction: Agrarian change, rural poverty and land reform in South Africa since 1994. *Journal of Agarian Change*, 13(1), 1-15.

Odden, M. & Wegge, P. (2005). Spacing and activity of leopards *Panthera pardus* in the Royal Bardia National Park, Nepal. *Wildlife Biology*, 11(2), 145-152.

Palmer, G., Birss, C., Kerley, G., Feely, J., Peinke, D. & Castley, G. (2017). *Cape Grysbok Raphicerus melanotis* [Online]. Retrieved May 25, 2019: https://www.iucnredlist.org/species/19306/50193334

Palmer, R. & Fairall, N. (1988). Caracal and African wildcat diet in the Karoo National Park and the implications thereof for hyrax. *South African Journal of Natural Science*, 18(1), 30-34.

Palomares, F. & Delibes, M. (1993a). Social organization in the Egyptian mongoose: Group size, spatial behavior and inter-individual contacts in adults. *Animal Behaviour*, 45, 917-925.

Palomares, F. & Delibes, M. (1993b). Key habitats for Egyptian mongooses in Donuana National Park, southwestern Spain. *Journal of Applied Ecology*, 30, 752-758.

Palomares, F. (2013). *Herpestes ichneumon* Egyptian mongoose (Ichneumon). In J. Kingdon & M. Hoffmann (Eds.). *The mammals of Africa, Volume V. Carnivores, pangolins, equids and rhinoceroses* (pp. 306-310). London: Bloomsbury Publishing.

Parsons, S., Smith, S.G.D., Martins, Q., Horsnell, W.G.C., Gous, T.A., Streicher, E.M. et al. (2008). Pulmonary infection due to the dassie bacillus (Mycobacterium tuberculosis complex sp.) in a free-living dassie (rock hyrax—*Procavia capensis*) from South Africa. *Tuberculosis*, 88(1), 80-83.

Peel, M.J.S. & Montagu, G.P. (1999). Modelling predator-prey interactions on a Northern Province game ranch. South African Journal of Wildlife Research, 29(2), 31–34.

Picker, M. & Griffiths, C. (2011). *Alien and invasive animals: A South African Perspective*. Cape Town: Struik Nature.

Pitman, R.T., Fattebert, J., Williams, S.T., Williams, K.S., Hill, R.A., Hunter, L.T. et al. (2017) Cats, connectivity and conservation: incorporating datasets and integrating scales for wildlife management. Journal of Applied Ecology, 54(6), 1687-1698.

Pool-Stanvliet, R. & Giliomee, J.H. (2013). A sustainable development model for the wine lands of the Western Cape: A case study of the Cape Winelands Biosphere Reserve. *AfriMAB*, 45-71.

Pool-Stanvliet, R. (2013). A history of UNESCO man and the biosphere programme in South Africa. *South African Journal of Science*, 109.

Pool-Stanvliet, R., Duffell-Canham, A., Pence, G. & Smart, R. (2017). *Western Cape biodiversity spatial plan handbook 2017.* Stellenbosch: Cape Nature.

Power, R.J. (2002). Prey selection of lions *Panthera leo* in a small, enclosed reserve. Koedoe, 45(2), 67-75.

Price, M.F. (2002). The periodic review of biosphere reserves: A mechanism to foster sites of excellence for conservation and sustainable development. *Environmental Science and Policy*, 5, 13-18.

Purves, M.G., Kruuk, H. & Nel, J.A.J. (1994). Crabs *Potamonautes perlatus* in the diet of otter *Aonyx capensis* and water mongoose *Atilax paludinosus* in a freshwater habitat. *Journal of Mammalogy*, 59, 332-341.

Quinn, R.D. (1986). Mammalian herbivory and resilience in Mediterranean-climate ecosystems. In Dell B., Hopkins A.J.M. & Lamont B.B. (Eds.), *Tasks for vegetation science, vol 16* (pp. 113-128). Dordrecht: Kluwer Academic Publishers.

Rabie, A. (2005). Biosphere reserves: The Kogelberg Biosphere Reserve. *Stellenbosch Law Review,* 16(1), 77-97.

Radloff, (2008). T*he ecology of large herbivores native to the coastal lowlands of the Fynbos Biome in the Western Cape, South Africa.* Unpublished doctoral dissertation, Stellenbosch University.

Radloff, F.G.T., Mucina, L., Bonad, W.J. & le Roux, P.J. (2010). Strontium isotope analyses of large herbivore habitat use in the Cape Fynbos region of South Africa. *Conservation Ecology*, 164, 567-578.

Rautenbach, T. (2010). *Assessing the diet of Cape leopard (Panthera paradus) in the Cederberg and Gamka mountains, South Africa.* Unpublished master's thesis, Nelson Mandela Metropolitan University.

Ray, J.C. (1997). Comparative ecology of two African forest mongooses, *Herpestes naso* and *Atilax paludinosus. African Journal of Ecology*, 35, 237-253.

Ray, J.C., Hunter, L. & Zigouris, J. (2005). Setting conservation and research priorities for larger African carnivores. *Wildlife Conservation Society*, 24.

Rebelo, A.J., Rebelo, A.G., Rebelo, A.D. & Bronner, G.N. (2019). Effects of alien pine plantations on small mammal community structure in a southern African biodiversity hotspot. *African Journal of Ecology*, 1-14.

Rebelo, A.G. (1992). Red data book species in the Cape Floristic Region: Threats, priorities and target species. *Transactions of the Royal Society of South Africa*, 48(1), 55-86.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M.C. (2006). Fynbos Biome. In L. Mucina & M.C. Rutherford (Eds.). *The Vegetation of South Africa, Lesotho and Swaziland* (pp. 52-219). Pretoria: Strelitzia 19.

Richardson, D.M., van Wilgen, B.W., Le Maitre, D.C., Higgens, K.B. & Forsyth, G.G. (1994). A computer-based system for fire management in the mountains of the Cape Province, South Africa. *International Journal of Wildland Fire*, 4(1), 17-32.

Ripple, W.J. & Beschta, J.A. (2004). Wolves and the ecology of fear: Can predation risk structure ecosystems? *BioScience, 54, 755-766*.

Ripple, W.J., Estes, J.A. Beschta, R.L., Wilmers, C.C., Richie, E.G., Hebblewhite, M. et al. (2014). Status and ecological effects of the world's largest carnivores. *Science*, 343, 151-162.

Roelke, M.E., Forester, D.J., Jacobson, E.R., Kollias, G.V., Scott, F.W., Barr, M.C. et al. (1993). Seroprevalence of infectious disease agents in free ranging Florida panthers (*Felis concolor coryi*). *Journal of Wildlife Diseases*, 29, 36-49.

Rouget, M., Richardson, D.M., Cowling, R.M., Lloyd, J.W. & Lombard, A.T. (2003). Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation*, 112, 63-85.

Rudel, T.K., Schneider, L., Uriarte, M., Turner II, B.L., DeFries, R., Lawrence, D. et al. (2009). Agricultural intensification and changes in cultivated areas, 1970–2005. *Proceedings of National Academy of Sciences*, 106(49), 20675-20680.

Sakai, A.K., Allendorf, F.W., Holt, J.S., Lodge, D.M., Molofsky, J., With, K.A. et al. (2001). The population biology of invasive species. *Annual Review of Ecology and Systematics*, 32, 305-332.

Sauvajot, R.M., Buechner, M., Kamradt, D.K. & Schonewald, C.M. (1998). Patterns of human disturbance and response by small mammals and birds in chaparral near urban development. *Urban Ecosystems*, 2, 279-297.

Schutte, A.L., Vlok, J.H.J. & Van Wyk, B.E. (1995). Fire-survival strategy - a character of taxonomic, ecological and evolutionary importance in fynbos legumes. *Plant Systematics and Evolution*, 195, 243-259.

Serieys, L.E.K., Bishop, J., Okes, N., Broadfield, J., Winterton, D.J., Poppenga, R.H. et al. (2019). Widespread anticoagulant poison exposure in predators in a rapidly growing South African city. *Science of the Total Environment*, 666, 581-590.

Schneider, M.F. (2001). Habitat loss, fragmentation and predator impact: Spatial implications for prey conservation. *Journal of Applied Ecology*, 38, 720-235.

Shipley, L.A. (1999). Grazers and browsers: How digestive morphology affects diet selection. In Launchbaugh, K.L., Sanders, K.D. & Mosley, J.C. (Eds.). *Grazing behavior of livestock and wildlife* (pp. 20-27). Moscow: Idaho Forest, Wildlife and Range Experimental Station Bulletin, 70.

Shiponeni, N.N. & Milton, S.J. (2005) Seed dispersal in the dung of large herbivores: Implications for restoration of Renosterveld shrubland old fields. *Biodiversity and Conservation*, 15, 3161-3175.

Silva, M., Hartling, L. & Opps, S.B. (2005). Small mammals in agricultural landscapes of Prince Edward Island (Canada): Effects of habitat characteristics at three different spatial scales. *Biological Conservation*, 126(4), 556-568.

Simberloff, D. (1998). Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? *Biological Conservation*, 83(3), 247-257.

Sinclair, A.R.E. (2003). The role of mammals as ecosystem landscapers. *Alces*, 39, 161-176.

Sinclair, A.R.E., Mertzger, K., Brashares, J.S., Nkwabi, A.G., Sharam, G. & Fryxell, J.M. (2010). Trophic cascades in tropical savannas: Serengeti as a case study. In J. Terborgh & J.A. Estes (Eds.), *Trophic cascades: Predators, prey, and the changing dynamics of nature* (pp. 255-274). Washington, DC: Island.

Skead, C.J. (1980). *Historical mammal incidence in the Cape Province, volume 1, The Western and Northern Cape*. Cape Town: Department of Nature & Environmental Conservation.

Skinner, J.D. & Chimimba, C.T. (2005). The mammals of the Southern African sub region. Cape Town: Cambridge University Press.

Skinner, J.D. & Smithers, R.H.N. (1990). *The mammals of the Southern African sub-region*. Pretoria: University of Pretoria Press.

Southey, D. (2009). Wildfires in the Cape Floristic Region: Exploring vegetation and weather as drivers of fire frequency. Unpublished master's thesis, University of Cape Town.

Stadler, H. (2006) Historical perspective on the development of problem animal management in the Cape Province. In B. Daly, W. Davies-Mostert, S. Evans, Y. Friedman, N. King, T. Snow et al. (Eds.), *Proceedings of a Workshop on Holistic Management of Human-Wildlife Conflict in the Agricultural Sector of South Africa* (pp. 11-16). Johannesburg: Endangered Wildlife Trust.

Statistics South Africa (2018). *Mid-year population estimates 2018, Statistical Release P0302*, Statistics South Africa, Pretoria.

Stein, A., Athreya, V., Gerngross, P., Balme G., Henschel, P., Karanth U. et al. (2016). *Leopard Panthera pardus* [Online]. Retrieved August 24, 2018: https://www.iucnredlist.org/species/15954/102421779

Stein, A.B. & Hayssen, V. (2013). *Panthera pardus* (*Carnivora: Felidae*). *Mammalian Species*, 45(900), 30-48.

Stevens, U. (2003). *The Winelands explorer*. Cape Town: Wanderlust Books.

Stuart, C., Stuart, T., & de Smet, K. J. (2013). African wildcat *Felis silvestris*. In J. S. Kingdon & M. Hoffmann (Eds.). *The mammals of Africa 5: Carnivora, pholidota, perissodactyla.* Amsterdam: Academic Press.

Stuart, C.T. & Stuart, M. (2015). *Stuart's field guide to mammals of Southern Africa*. Cape Town: Struik Nature.

Stuart, C.T., Macdonald, I.A.W. & Mills, M.G.L. (1985). History, current status and conservation of large mammalian predators in Cape Province, Republic of South Africa. *Biological Conservation*, 31, 7-19.

Stuart, C.T., Stuart, T. & Pereboom, V. (2003). Diet of bat-eared fox (*Otocyon megalotis*), based on scat analysis, on the Western Escarpment, South Africa. *Canid News*, 6(2).

Sturtevant, B.R., Miranda, B.R., Yang, J., He, H.S., Gustafson, E.J. & Scheller, R.M. (2009). Studying fire mitigation strategies in multi-ownership landscapes: Balancing the management of fire-dependent ecosystems and fire risk. *Ecosystems*, 12, 445-461.

Swanepoel, L.H., Balme, G., Williams, S., Power, R.J., Snyman, A., Gaigher, I. et al. (2016). A conservation assessment of *Panthera pardus*. In M.F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo & H.T. Davies-Mostert (Eds.), *the red list of mammals of South Africa, Lesotho and Swaziland* (pp. 1-13). Pretoria: South African National Biodiversity Institute & Johannesburg: Endangered Wildlife Trust.

Swanepoel, L.H., Lindsey, P., Somers, M.J., van Hoven, W. & Dalerum F (2013). Extent and fragmentation of suitable leopard habitat in South Africa. Animal Conservation, 16, 41-50.

Syphard, A.D., Radeloff, V.C., Hawbaker, T.J. & Stewart, S.I. (2009). Conservation threats due to human-caused increases in fire frequency in Mediterranean-climate ecosystems. *Conservation Biology*, 23(3), 758-769.

Syphard, A.D., Radeloff, V.C., Keeley, J.E., Hawbaker, T.J., Clayton, M.K., Stewart, S.I. et al. (2007). Human influence on California fire regimes. *Ecological Applications*, 17(5), 1388-1402.

Taylor, A. & Lehmann, T. (2015). *Orycteropus afer Aardvark* [Online]. Retrieved from April 27, 2019: https://www.iucnredlist.org/species/41504/21286437

Taylor, A., Cowell, C. & Drouilly, M. (2017). *Grey rhebok Pelea capreolus* [Online]. Retrieved April 27, 2019: from: https://www.iucnredlist.org/species/16484/50192715

Tizora, P., Le Roux, A., Mans, G. & Cooper, A. (2016). *Land use and land cover change in the Western Cape Province: Quantification of changes & understanding of driving factors.* Unpublished paper delivered at the Seventh Planning Africa Conference on Making Sense of the Future: Disruption and Reinvention. Johannesburg, July, 4-6.

Tizora, P., Le Roux, A., Mans, G. & Cooper, A. (2018). Adapting the Dyna-CLUE model for simulating land use and land cover change in the Western Cape Province. *South African Journal of Geomatics*, 7(2), 190-203.

Van der Zee, D. (1981). Density of Cape clawless otters *Aonyx capensis* (Schinz, 1821) in the Tsitsikama Coastal National Park. *South African Journal of Wildlife Research*, 12(1), 8-13.

Van der Zyl, D.W. & Kruger, F.J. (1975). Results of the multiple catchment experiments at the Jonkershoek Research Station, South Africa. II. Influence of protection of fynbos on stream discharge in Langrivier. *Forestry in South Africa*, 16, 13-18.

Van Hensbergen, H.J., Botha, S.A., Forsyth, G.G. & Le Maitre, D.C. (1992). *Do small mammals govern vegetation recovery after fire in fynbos? Fire in South African Mountain Fynbos*. Berlin: Springer-Verlag.

Van Wilgen, B.W. (1987). Fire regimes in the Fynbos biome. In R.M. Cowling, D.C. Le Maitre, B. McKenzie, R.P. Prys-Jones & B.W. van Wilgen (Eds.), *Disturbance and the dynamics of Fynbos Biome communities* (pp. 6-12). Pretoria: Foundation for Research Development.

Van Wilgen, B.W. (2009). The evolution of fire and invasive alien plant management practices in fynbos. *South African Journal of Science*, 105, 335-342.

Van Wilgen, B.W. (2013). Fire management in species-rich Cape fynbos shrublands. *Frontiers in Ecology and the Environment*, 11(1), 35-44.

Van Wilgen, B.W., Forsyth, G.G., de Klerk, H., Das, S., Khuluse, S. & Schmitz, P. (2010). Fire-management in Mediterranean-climate shrublands: A case study from the Cape Fynbos, South Africa. *Journal of Applied Ecology*, 47, 631-638.

Van Wilgen, B.W. & Scott, D.F. (2001). The effect of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: A simulation study. *Journal of Applied Ecology*, 22, 955-966.

Venter, J., Ehlers-Smith, T. & Seydeck, A. (2016). A conservation assessment of *Potamochoerus larvatus.* In M.F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo & H.T. Davies-Mostert HT (Eds.), *The Red List of Mammals of South Africa, Swaziland and Lesotho.* South African National Biodiversity Institute and Endangered Wildlife Trust, South Africa.

Vlok, J.H.J. & Yeaton, R.I. (2000). The effect of short fire cycles on the cover and density of understorey sprouting species in South African mountain fynbos. *Diversity and Distributions,* 6, 233-242.

Wang, S.W. & Macdonald, D.W. (2009). The use of camera traps for estimating tiger and leopard populations in the high altitude mountains of Bhutan. *Biological Conservation*, 142, 606-613.

Whittington-Jones, G.M., Bernard, R.T.F. & Parker, D.M. (2011). Aardvark burrows: A potential resource for animals in arid and semi-arid environments. *African Zoology*, 46(2), 362-370.

Wilcove, D.S., Rothstein, D., Dubow, D., Phillips, A. & Losos, E. (1998). Quantifying threats to imperiled species in the United States. *BioScience*, 48, 607–615.

Willan, K. & Bigalke, R.C. (1981). The effects of fire regime on small mammals in S.W. Cape Montane Fynbos (Cape Macchia). In C.E. Conrad & W.C. Oechel (Eds.), *Dynamics and Management of Mediterranean-type Ecosystems* (pp. 207-212). San Diego: San Diego State University.

Wimberger, K., Downs, C.T. & Perrin, M.R. (2009). Two unsuccessful reintroduction attempts of rock hyraxes (*Procavia capensis*) into a reserve in KwaZulu-Natal Province, South Africa. *South African Journal of Wildlife Research*, 39(2), 192-201.

Wolf, C. & Ripple, W.J. (2016). Prey depletion as a threat to the world's large carnivores. *Royal Society Open Science*, 3:160252.

Woodroffe, R. & Ginsberg, J.R. (1998). Edge effects and extinction of populations inside protected areas. *Science*, 280(5372). 2126-2128.

Young, J.K., Olsen, K.A., Reading, R.P., Amgalanbaatar, S. & Berger J. (2011). Is wildlife going to the dogs? Impacts of feral and free-roaming dogs on wildlife. *Bioscience*, 61(2), 125-132.

Appendix 1.1: List of study species, their scientific and other name

Name	Scientific name
Leopard	*Panthera pardus pardus*
Caracal	*Caracal caracal*
African wild cat	*Felis silvestris lybica*
Feral domestic cat	*Felis catus*
Cape fox	*Vulpus chama*
Bat-eared fox	*Otocyon megalotis*
Aardwolf	*Proteles cristata*
Feral domestic dog	*Canis lupus familiaris*
Honey badger	*Mellivora capensis*
Water mongoose	*Atilax paludinosus*
Large grey mongoose	*Herpestes ichneumon*
Cape grey mongoose	*Galerella pulverulenta*
Large OR small spotted genet	*Genetta tigrine/ Genetta genetta*
Cape clawless otter	*Aonyx capensis*
African striped weasel	*Poecilogale albinucha*
Striped polecat	*Ictonyx striatus*
Chacma baboon	*Papio ursinus*
Aardvark	*Orycteropus afer*
Bush pig	Potamochoerus larvatus
Feral pig	*Sus scrofa*
Cape grysbok	*Raphicerus melanotis*
Klipspringer	*Oreotragus oreotragus*
Common duiker	*Sylvicapra grimmia*
Grey rhebok	*Pelea capreolus*
Rock hyrax	*Procavia capensis*
Bushbuck	*Tragelaphus sylvaticus*
Cape porcupine	*Hystrix africaeaustralis*
Cape OR scrub hare	Lepus capensis/ *Lepus saxatilis*
Smith's red rock rabbit	*Pronolagus rupestris*

Chapter 2

Land-cover and fire regime changes in the Boland Mountain Complex

Brittany C. Schultz[1], Anita Wilkinson[2], Alison J. Leslie[1]

[1]Department of Conservation Ecology and Entomology, University of Stellenbosch, Matieland 7602, Western Cape, South Africa.

[2]The Cape Leopard Trust, P.O. Box Box 31139, Tokai, 7966, Cape Town, South Africa.

2.1 Abstract

Medium-sized mammals face range declines throughout South Africa, which are driven by land-cover change from urbanisation and agricultural expansion. Changes to fire regimes as disturbances are a further concern for natural habitats' ability to support current medium-sized mammal population abundances. Both these changes, that are often anthropogenic effects, are threats for medium-sized mammal populations' survival in the Boland Mountain Complex (BMC). Recorded leopard diet in the BMC reflects multiple medium-sized mammal species. The BMC forms a key part of leopard ranges within the Fynbos biome. The study area contains core protected areas (PA), is surrounded by agricultural and plantation buffer areas and is influenced by multiple built-up towns, that have increasing development rates and population sizes. This study aimed to determine if and what significant loss of natural habitat has occurred through land-cover change and a shift in fire regime patterns from 1957 to 2017, with regards to habitat availability for leopard and medium-sized mammals. This study utilised Geographic Information Systems (GIS) to analyse the South African National Land-cover shapefiles from 1990 and 2013 to quantify land-cover change. The shapefiles of CapeNature's historic fire records were used to quantify fire regime changes from 1957 to 2017. There was a total gain of 107km^2 vegetation land-cover. Plantation conversions to vegetation were the largest land-cover shift displayed in the landscape. The encroachment of agriculture and built-up areas into key functional habitats, such as corridors within the BMC and into external leopard ranges raised some concerns. Fire regimes showed increases in fire frequencies from 1957 to 2017. Greater areas of land were burning per year from 1987 to 2017 than from 1957 to 1987, as a result of more, smaller fires. An increase in human ignition sources are the likely cause for this. By 2002 fire regimes had become homogenous, with near equivalent Fire Return Intervals (FRI) throughout the landscape, from patchy, heterogenous FRI's with great variations between Biosphere Reserves (BR) and among Mountain Zones (MZ) between 1957 and 1987. In terms of land-cover change, sufficient sized areas of vegetation remained intact and have replaced anthropogenic land-covers, to act as mammalian habitat with fewer negative human anthropogenic land-cover impacts. Changes to fire regimes, however, raise concerns for mammal's access to safe refuges and adequate habitat resources in vegetation after natural reestablishment. Both human influences reinforce the importance of landscape planning and management of natural corridors and isolated habitats.

Key words**:** Fire regimes, Fynbos Biome, Geographic Information Systems, Habitat loss, Land-cover change, Mammals.

2.2 Introduction

Medium-sized mammals are currently facing large range declines globally and throughout South Africa, primarily attributed to habitat loss, fragmentation, agricultural expansion and urbanisation (Fairbanks et al., 2004; Hoffman et al., 2010; Jacobson et al., 2016; Ray et al., 2005). Human development and agricultural expansion are the biggest recognised threats to natural habitats on a global scale (Hansen & DeFries, 2017; Kintz et al., 2006). The resultant habitat losses from land-cover change, combined with changes to natural disturbances, such as fire, can greatly impact mammal ranges (Bradstock, 2008; Griffiths & Brook, 2014a). Natural fire regimes in some parts of the world, have experienced changes from human development and land-cover change that likely negatively impact the natural ecology (Clarke, 2008; Cochrane & Laurance, 2004; Syphard et al., 2009).

Loss of habitat is driven by shifts in land-covers. Reductions in the size of natural habitat available and a change in the functionality of areas affect many mammals' survival capabilities. Generally, farms and plantations have lower diversities and abundances of mammal species than neighbouring natural habitats that further decline with the intensification of these human land-uses (Benton et al., 2003; Tscharntke et al., 2005; Zanne et al., 2001). Agriculture and plantation land-covers within buffer edges can accommodate and further benefit various mammal species, acting as extended habitat and movement corridors (Henzi et al., 2011; Tscharntke et al., 2005; Zanne et al., 2001), although, functionality of corridors usually decreases with anthropogenic land-cover intensification (Vogt et al., 2009). Other mammal species may however be unable to utilise agriculture and plantation land-covers and the natural vegetation these human moderated landscapes replaced is thus habitat lost (Brodie et al., 2014; DeFries et al., 2007). As a result, mammal population sizes may decline in the subsequently reduced remaining available habitat (Fahrig, 1997).

Land-covers burn differently to one another and some; such as forestry, infrastructure or various crops, can act as stronger fuels than natural vegetation, increasing fire intensity, thus resulting in more permanent habitat damage or a higher number of mammal deaths (Cochrane & Laurance, 2002). Agricultural and developed land-covers (industrial/residential for example) may provide more possible sources of fire ignitions, which may increase fire frequency (Archibald et al., 2012). The development and expansion of urban areas and agriculture, fragments and limits refuges into which species may flee and causes direct mortalities (Bradstock, 2008; van Wilgen et al., 2010; Keeley et al., 2011; Krebs et al., 2010). When such fragmented habitat burns, the burnt area is temporarily unable to support species to the same extent as previously and can result in increased resource competition (Converse et al., 2006; Griffiths & Brook, 2014b). Too frequent fires or high burn intensities alter vegetation structure and floral species composition, and thus negatively impact the ecosystem and mammals in it (Griffiths & Brook, 2014b; Rabinowitz, 1990; Rowe-Rowe & Lowry, 1981; Whelan, 1995). On the other hand, Keyser & Ford (2006) found fire exclusion in forest ecosystems can be a greater threat to some mammal species. A shift of fire frequency, in either direction, has negative consequences. It therefore is necessary to assess spatial and temporal fire regimes in relation to available habitat in the BMC (Legge et al., 2008).

Many species can experience short-term benefits from a recently burnt landscape (Converse et al., 2006; Griffiths & Brook, 2014b; Parrini & Owen-Smith, 2009). Herbivores often consume the remaining ash and many species show a preference for the initial green shoots over established vegetation (Komarek, 1969; Parrini & Owen-Smith, 2009). Grey rhebok (which are predated on by leopard) have been documented gathering in recently burnt plains, in nearby reserves (Beukes, 1987; Novellie, 1987). Insectivores are supported by influxes of insects to new vegetation growth and predators take advantage of the low vegetation cover that results in prey being more visible (Lawrence, 1966; Rowe-Rowe & Lowry, 1981). This advantage to predators may negatively impact prey species as they are more susceptible to being hunted in burnt habitats (Birtsas et al., 2012; Lawrence, 1966). Dingoes *Canis lupis dingo* in Austrailia have shown a preference to the limited vegetation cover after a fire, which Geary et al. (2018) believed may ease their role as apex predators of predating on mesopredators (Hradsky et al., 2017). This reaction to burnt habitats has not officially been documented in leopard but may also be true for them as apex predators within the BMC. Chia et al. (2016) found variation amongst different mammalian species' reactions to how fires burn in a landscape and that heterogenous patterns are important for multiple species to persist.

The term "habitat" is broadly defined and can be narrowed down to the basic factors of its location, size and the resources contained therein (Hall et al., 1997). Leopard in the BMC are the apex predators in the region with comparatively large home range requirements (from 100km^2 up to 600km^2) (Boshoff et al., 2002; Kerley et al., 2003; Martins & Martins, 2006; Stein et al., 2016). Thus, they are an ideal umbrella species when considering areas of viable habitat for medium-sized mammals (Boshoff et al., 2002; Crooks, 2002; Martins & Martins, 2006; Stein et al., 2016). Quantifying how humans have affected the size of available mammalian habitat in the study area is a good initial measurement to determine whether there are increasing threats to leopard and therefore all other medium-sized mammals.

Human population growth is the root of land-cover and fire regime changes in the last three centuries in the Western Cape (Kraaj & van Wilgen, 2014; Tizora et al., 2016; van Wilgen et al., 2010). This province has a current human population size of over 6.5 million and has had a high growth rate (11.49% annually) since 2011 (Spatial Planning, 2018; Statistics South

Africa, 2018). By 2040 the population is expected to increase to 7.36 million people, (a growth rate of 13.23%) (Spatial Planning, 2018; Statistics South Africa, 2018). This growth is attributed to a natural population increase and migration of people into areas of high economic boom, thus improving the development potential of local towns (Donaldson et al. 2012; Tizora et al., 2016, 2018). Tizora et al. (2016) found that the most significant increase in infrastructure expansion in the Western Cape was in the Cape Winelands area (which falls within the BMC). This expansion encroached on agriculture, forestry and vegetation land-covers. Further, Hannah et al. (2013) predicted that elevated areas within the Cape Winelands are at risk of conversions from fynbos into viticulture land-cover. Fairbanks et al. (2004) too, predicted major shifts and growth of viticulture over key ecosystems in the Fynbos biome. These localised expansions are of concern as to how land-cover changes may affect mammalian habitat in the BMC (Tizora et al., 2016).

Fire is a key abiotic phenomenon in the Fynbos biome that helped drive the high ecosystem biodiversity, and many local flora and fauna are adapted to a specific natural fire regime (Bowman et al., 2011; Forsyth & van Wilgen, 2008; Keeley, 2002; Keeley et al., 2011; Kraaij & van Wilgen, 2014; van Wilgen et al., 2010). The fire ecology in the Cape concurs with many of the Mediterranean ecosystems globally, showing an increase in the frequencies and sizes of wildfires (Archibald et al., 2012; Forsyth & van Wilgen, 2008; Syphard et al., 2009). A general maximum period that fynbos plant species can withstand no burning (before some species go into senescence) is 30-years (van Wilgen, 2009). Southey (2009) has highlighted increased fire ignitions in the Hottentots-Holland Nature Reserve and Gumbi (2011) found the Kogelberg Nature Reserve's fire regime increased in frequency from 1980 to 2006, and that patches burn an average of every 15-years. Most local fire ecology research has focused on landscape patterns, the effects on vegetation, and a few on small mammals roles in floral survival or large mammals roles in vegetation disturbance (Auld & Denham, 2001; Forsyth & van Wilgen, 2008; Gumbi, 2011; van Hensbergen et al., 1992; van Wilgen et al., 2010; Willan & Bigalke, 1981). Research on medium-sized mammals and fire is particularly data-deficient for the Fynbos biome (Bowman et al., 2011; Lindenmayer et al., 2016).

The natural fynbos vegetation in the BMC is highly susceptible to fires and many of the invasive alien species burn easily and more intensely (Kraaij et al., 2011; Mack & D'Antonia, 1998). The close proximity of these vegetation types to the above-mentioned human sources of fire, as well as the impacts of climate change (causing higher average, local temperatures and drought) may have some severe impacts on the natural fire regime of the study area's buffer zones (Archibald et al., 2012; Gumbi, 2011).

PAs in the BMC are under jurisdiction of the National Environmental Management: Protected Areas Act (NEM:PAA, 2003) (Pool-Stanvliet et al., 2017). This act is enforced by CapeNature, the Western Cape's conservation body that is responsible for guiding/monitoring land-cover changes and human encroachment (Pool-Stanvliet et al., 2017). Buffer zones are of major importance to PAs (Pool-Stanvliet et al., 2017). The World Wide Fund for Nature (WWF) and CapeNature initiated the Conservation Stewardship and Conservation Champions programmes respectively, to ensure sustainable practices on some of the private buffer properties in 2003 (Hannah et al., 2013). There remains the potential risk of the land-owners developing further into buffer areas (Hansen & DeFries, 2007; Pool-Stanvliet et al., 2017). Built-up landcovers are often controlled by various private and government stakeholders, depending on the allocated land-zoning and governed by the National Environmental Management; Integrated Coastal Management Act 76 (NEMICMA) (Wylie, 2016). These built-up/developed landcovers contain residential, informal settlements, industrial and urban land-uses (Ishwaran et al., 2008). Therefore, they are generally unable to support medium-sized mammals, often resulting in complete habitat loss and can have strong negative impacts on surrounding habitats (Armstrong et al., 1996). The buffer zone land-covers comprise of various agricultural practices (mostly plant-based), pine plantations and some private nature reserves. Core zones contain natural protected habitats (Ishwaran et al., 2008). Natural vegetation in the core and buffer zones should act as functional habitats. These habitats, when of the correct size and containing the necessary resources, should support full diversities of extant medium-sized mammals (Hall et al., 1997).

This study examined land-cover change using the same 1990 and 2013/2014 South African Land-cover datasets that Tizora et al. (2016) used to examine the entire Western Cape's land-cover changes. By using the same datasets, the current study examines the authors' described land-cover changes on a smaller-scale. This allows for a more localised understanding of causes and more accountability for actions that drive them. The historic fire records from the South African National Biodiversity Institute (SANBI) have been utilised in many previous studies across the Western Cape to analyse various aspects of fire regimes. These studies include Gumbi (2011) in the Kogelberg Nature Reserve and Southey (2009) in the Hottentots-Holland Nature Reserve, which are located within the study area. Using the same dataset enables accurate comparisons to what has already been observed in the study area. A study which incorporates localised spatial and temporal impacts, is needed to truly understand the unique, location-specific landscape patterns and what mammalian habitat may remain (Fraser, 1990; Gumbi, 2011; Lindenmayer et al., 2016; Richardson et al., 1994; Schwilk et al., 1997; Syphard et al., 2009).

The Boland Mountain Complex (BMC) (Figure. 2.1) is one of eight protected United Nations Environmental, Educational, Scientific and Cultural Organization's (UNESCO) World Heritage sites of the Fynbos biome in the Western Cape Province, South Africa. The BMC spans over two UNESCO Biosphere Reserves: the Kogelberg Biosphere Reserve (KBR) and the Cape Winelands Biosphere Reserve (CWBR) and includes the Limietberg, Jonkershoek, Hottentots-Holland and Kogelberg Nature Reserves. Leopard and medium-sized mammals range within the BMC's section of the Cape Fold mountains and surrounding buffer land-covers (see Chapter 1). The site is considered a key protected portion of these mammalian populations (Mann et al., 2019).

The objective of this study was to assess and illustrate how much area of potential medium-sized mammalian habitat is still available in the BMC and how this has changed in the 60-year period from 1957 to 2017. The primary aim of this study was to quantify and determine whether significant natural habitat loss has occurred through land-cover change and a shift in fire regime patterns. Primary objectives:

i) To quantify what and identify where land-cover and fire regime changes have occurred and what key habitats may be under threat.

ii) To determine if significant variances in land-cover changes and fire regime patterns exist between the two biosphere reserves and seven mountain zones.

iii) To determine what factors in the landscape may be influencing these changes

iv) To determine whether changes in habitat size from land-cover shifts and fire mismanagement may pose future threats to medium-sized mammals.

v) To provide management recommendations.

2.3 Methodology

2.3.1 Study area and sites

The study area covers a total of 3635km^2. The area is focussed around the BMC and its ranges that are utilised by local leopard populations This incorporates entire core sections, an extent of the buffer land-covers that surround cores and partial sections of transitional zones, as appropriate for the aims of the study. The study area is influenced by 18 towns and was divided into seven "Mountain Zones" (MZ) (East Hawequas, West Hawequas, Simonsberg, West Hottentots-Holland, Theewaterskloof Basin, Groenlandberg and Kogelberg MZs) (Figure. 2.1). Vegetation corridors that continue into other natural ranges (Groot Winterhoek Mountains, Riviersondereind Nature Reserve and the coastal edge leading east toward Walker Bay area) were excluded at the biosphere reserve borders. The study areas border extends into areas of natural habitat that continue a short distance before becoming transformed land-uses. Transformed zones and land-uses were cut-off where assumed unsuitable to the study species (Wilkinson, personal communication, 2017). All land-cover changes in this study thus reflected the inward changes toward the BMC's mammalian habitats, and not changes to external habitats outside of the study area (Figure 2.1).

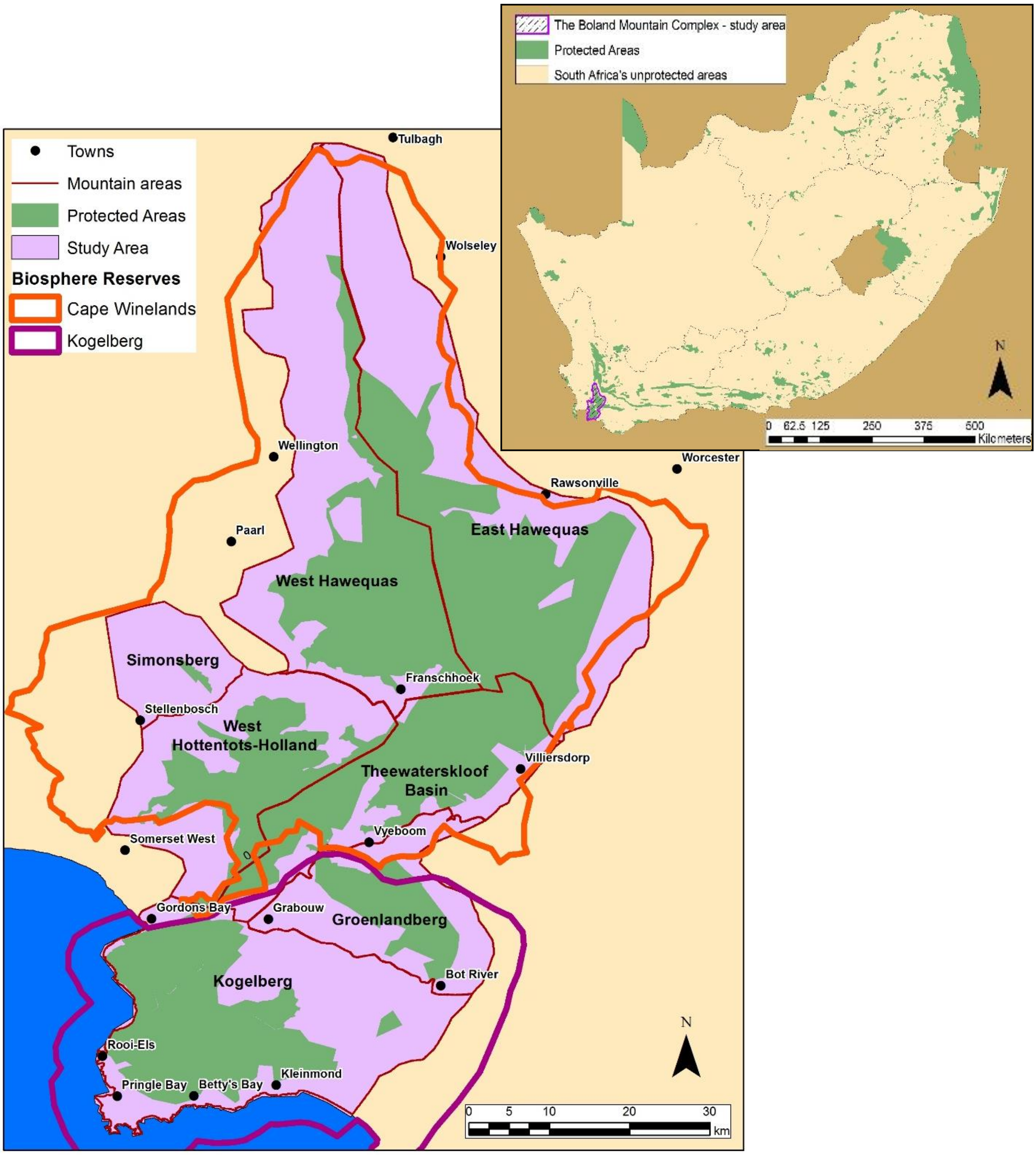


Figure 2.1: Location of the focal study area within the Boland Mountain Complex location in the Western Cape Province, South Africa and its two biosphere reserves and seven mountain zone divisions amongst protected areas and towns.

2.3.2 Data Sources

Land-cover data

The land-cover data layers were sourced from and created by GEOTERRAIMAGE (GTI) and licensed to the South African Department of Environmental Affairs (DEA). Landsat images were sourced from the United States Geological Survey (http://glovis.usgs.gov/) (WGS 84 / UTM zone 35S projection and coordinate reference system) and used to create land-cover datasets for two time periods, namely 1990 and 2013/14. A 1990 South African Land-cover dataset was created from Landsat 4 and 5, after the 2013/14 South African National Land-cover dataset was created from Landsat 8 imagery. Both land-cover datasets were produced as 72-class raster layers, with 30mx30m cells by a semi-automated modelling procedure and were then further reclassified into 35-classes, for this study. The Normalised Difference Vegetation Index (NDVI), Normalised Difference Water Index (NDWI) and GTI's in-house algorithms were spectral indices used over different seasons and landscapes to produce the classes (GEOTERRAIMAGE, 2014b).

GEOTERRAIMAGE (2014b) ran accuracy assessments for only the 2013/14 land-cover datasets as no references were available for 1990, however it was assumed they reflect adequate accuracy for 1990 because the same mapping and modelling procedures were performed. GEOTERRAIMAGE (2014b) found that the 2013/14 classes showed a mean accuracy across the land-covers of 88.36%. A Kappa Index value of 80.87% was scored indicating the results were unlikely due to chance. User and producer accuracy tests were run per class to assess for inaccuracies. Scale and level of detail being analysed is also important to understand as shifts occur within a generalised land-cover (GEOTERRAIMAGE, 2014a).

Fire data

The historic fire records were sourced from CapeNature's database of fires that took place in CapeNature's protected areas and state-owned forests, from 1958 to recent. These records belong to CapeNature and were most recently updated in July 2017. Individual fire records were then digitized into multiple polygons in one shape file from previous reports, fire scars, and aerial photographs since 1998, and satellite images and GPS data sourced from aerial and ground-level fire line assessments. Attribute data (such as fire sources, sizes, reserves and locations) were sourced and added from the central fire database using QGIS. The final product had been regularly updated and confirmed in 2005 using SPOT5 satellite mosaic images. Fires were generally only recorded in the now core protected areas and many of the state forests. All conclusions drawn on fire regimes refer to protected habitat, numerous state forests and some agricultural areas, because it was generally in CapeNature managed areas where fires were recorded. Fires analysed in this study thus reflect mostly fires occurring in vegetation and forestry land-covers.

2.3.3 Methods

ArcMap 10.4.1. (ESRI, 2015) was the predominant GIS programme used to quantify both the land-cover changes and fire frequencies, as well as QGIS to assign co-ordinate systems to new layers. ArcMap and Microsoft excel (2016) were used to produce map and graph results. Statistical analyses were run on both IBM's SPSS Statistics (2017) and Statistica version 13.3. (TIBC Software Inc, 2017).

Land-cover changes

The 35 original classes from the DEA's 1990 and 2013/2014 land-cover shape files were reclassified into five general classes with ArcMap 10.4.1 (ESRI, 2015), namely "vegetation", "plantation", "agriculture", "built-up" and "water" (Table 2.1). For this study the 2013/14 land-cover dataset was said to represent the year 2013. The five general classes were decided based on their assumed accessibility as habitats for all medium-sized mammals. All vegetation types, regardless of structure, origin or species were grouped into the "vegetation" class that represented habitat and land-covers that leopard were able to fully utilise. This group was regarded most appropriate as some of the vegetation-type classes in the original dataset, scored low accuracy results and in some areas their vegetation structures and types were in-differentiable from other classes. For example, the class "bush and thicket" was often noted as indifferentiable from invasive species, and "fynbos low shrub" was often not distinct from "other low shrubs" (GEOTERRAIMAGE, 2014b). "Indigenous forest", "woodland", and "thicket" scored low User Accuracies of less than 75% (GEOTERRAIMAGE, 2014b). Agriculture and built-up land-covers were left as broad classes because cultivated lands subsets would often change quite substantially between seasons, and built-up subclasses were based on "typical" building characteristics and not accurately defined. The "agriculture" class included all forms of agriculture recognised in the Landsat images, consisting mostly of cultivated orchards, vineyards and annual crops, including subsistence farming. "Built-up" land-cover classes referred to settlements and mines due to the potential of mines having large infrastructure and being non-ideal leopard habitat. GTI's definition of settlements includes residential, urban and informal settlements, industrial land-cover and other built/constructed environments. The class "plantation" contained the same single plantation class that the data originally contained and refers to any areas of adult, juvenile and felled plantations as well as windbreaks. "Water" was a class of both open bodies of water and wetlands. Open bodies refer to natural rivers, lagoons, lakes and the ocean as well as man-made dams (GEOTERRAIMAGE, 2014a).

Because natural water pools and man-made dams were not differentiable in the original classes, they were grouped together with wetlands as a water supply in the study area.

Table 2.1: Original names of land-cover classes by the Department of Environmental Affairs land-cover shape files from 1990 and 2013 and their reclassified names used in this study

Reclassified land-cover	Original land-cover name
Water	Permanent water
	Seasonal water
	Wetland
Vegetation	Indigenous forest
	Dense bush, thicket & tall dense shrubs
	Woodland and open bushland
	Grassland
	Low shrubland: fynbos
	Low shrubland: other
	Erosion dongas and gullies
	Bare (non-vegetated)
Agriculture	Commercial annuals
	Commercial pivot
	Commercial permanent (orchards/vines)
	Subsistence
Forestry	Forest plantations: mature trees
	Forest plantations: young trees
	Forest plantations: temporary clear-felled stands
Built-up	Mine (1) bare
	Mine (2) semi-bare
	Mine water seasonal
	Mine water permanent
	Mine buildings
	Commercial
	Industrial
	Informal
	Residential
	Schools and sports grounds
	Smallholding
	Sports and Golf
	Township
	Village
	Built-up

The relevant data for the study area were clipped from the reclassified South African 1990 and 2013 land-cover data layers using a line shape file. Both years land-cover raster layers were properly aligned with one another's edges using a spatial adjustment and all recorded changes were a true reflection. The area covered per period for each land-cover was calculated and displayed using ArcMap and Microsoft's Excel 2016. Change in area covered was documented on three scales, that of the whole study area, the two biosphere reserves and the seven defined major MZs.

The 2013 dataset was overlaid on the 1990 dataset to display and calculate the gain, loss and persistence area for each land-cover class (Manandhar et al., 2009). A matrix of areas of land-cover gain, loss and persistence were created using the "tabulate area" feature on ArcMap 10.4.2, with the following calculations. The first column contained the class names for each land-cover of 1990, and across the first row each column is headed with the class names for 2013. All the values which occurred on the diagonal intersection from the top left corner to the bottom right were areas for that class which persisted with un-changed land-cover from 1990 through 2013. The total for each class in 2013 was visible at the end of each column and the totals for 1990 were at the end of each row. Gross gain was in the last row and calculated by subtracting class persistence from the 2013 class (column) total. Gross loss was the next column and was the result of subtracting class persistence from the 1990 class (row) total. Total change was the gross gain added to gross loss, net change equals to gross gain minus gross loss and swap change is total change minus net change.

Fire Mapping

CapeNature's fire database contains records that date back as far as 1945, however records for the early period appeared incredibly sparse. The reason for this is unknown but may be due to fewer fires or low recording rates at the time. For the purpose of the current analyses, the fire layer was classified into distinct fire years. A fire year runs from June of one year to July of the next year, based on the fire season in the Western Cape running from October to March (van Wilgen, 2009). Fire data for each specified fire year were exported from the original SANBI fire data-layer into separate shape files of each fire year. Based on Gumbi (2011), who concluded the most recent fire frequencies in the Kogelberg Nature Reserve (within the study area) to be about 15-years, the fire years were grouped into four 15-year periods – including data from 1957 to June 2017. This 15-year period is also a factor of 30, allowing the periods to be assessed according to the 30-year period that Forsyth and van Wilgen (2008) described as the maximum period which many Fynbos species can survive without burning.

By merging all individual fire polygons per fire year, the total burn-scar area was calculated for each fire year. The number of fires per year was counted as each appeared separately in the attribute table using Microsoft Excel (2016). The attributes of all fires that had occurred over each 15-year period were merged, as mentioned above, into four 15-year period data layers and the total area burnt for each of the five land-cover classes per year was subsequently calculated in ArcMap. Each of the four 15-year period fire data layers was clipped to produce a separate layer for each of the seven MZs, resulting ultimately in 28 additional 15-year fire period data layers. The area burnt for each of the five land-cover classes for each of the seven MZs during each of the four 15-year periods was then calculated. The total area burnt and number of fires of each fire year were represented as a statistical sample within the 15-year periods of 1957 to 1972 (period A), 1972 to 1987 (period B), 1987 to 2002 (period C) and 2002 to 2017 (period D).

Using Chao's (2002) Fire Return Interval (FRI) formula, the mean number of years per period to burn was calculated at three spatial scales, i.e.: at the study area, biosphere reserve and MZ level.

$FRI = T \times \frac{A}{B}$ equates to the number of years on average for an area to burn.

T is the number of years that the interval is being assessed along, A is the size of the focus area in hectares and B is the number of hectares that burnt during the focus period.

At both the study area and biosphere level FRI was calculated over the four 15-year periods and included settlements and farms for which the fire data were not reported. This was done to illustrate a comparable figure of the BMC to the biosphere reserves and included a section that had not yet burnt, even though some were incapable of burning. On the MZ level, the FRI for each MZ over the four periods was calculated with *A* defined as only the areas which had been burnt in the total 60-year study period.

The fire frequencies across the study area and entire study period were calculated and displayed by converting all the selected periods' individual fire shape files into raster data. Each class represented the number of burns contained in the majority of each 30mx30m cell. This was done by making a union of the individual fire polygons and calculating each central geographic location. Each polygon location was compared and summarised into a table which was then joined to the dissolved shape file and finally converted to a raster file to produce Figure 2.7. Prior to rasterization, stratified random samplings were used to select one-thousand of the polygons (representing the count of fires) created. Settlement polygons were sourced from the 2013/2014 DEA's land-cover raster file, roads were those of South Africa and Lesotho, sourced from Open Street Map (2014) and the distances to these were measured using the near tools geodesic method.

2.3.4 Data Analysis

Land-cover

Paired T-tests and Wilcoxon Signed Rank Tests were run to detect whether the percentages of each land-cover class differed between 1990 and 2013 across all seven of the MZs. Percentages for each MZ were treated as variables and thus N=7 (McKillup, 2005). After a Shapiro-Wilk test located significant outliers in the data set, they were removed. The test was repeated, and data were proved normal for all classes except for vegetation. A paired t-test was then run to test for significant mean differences from 1990 to 2013. A Wilcoxon Signed Rank Test was used as a non-parametric t-test and analysed all variables for the non-normal vegetation class, including the outliers.

Fire Regime

Levene's tests were performed in SPSS (2017) to test the homogeneity of both the number of fires per year and the land-cover burnt per year. Welch's ANOVAs were run when the homogenate variance assumption was not met, and a Games Howell post hoc test then distinguished where the significant differences lay. To test for significance between the rank-distribution of the FRIs over the four time periods for all seven MZs, Friedman's test was used. The Wilcoxon Signed Rank Test was run as a post hoc with a Bonferroni adjustment. Pearson's correlation was run on the normally distributed number of fire data and the distances from human settlements and major roads (McKillup, 2005).

2.4 Results

2.4.1 Quantifying land-cover change

Land-cover class composition and changes

Vegetation made up most of the study area in both 1990 (72.70%) and 2013 (75.70%), agriculture accounted for the second largest area for both years (14.20% to 14.40%). Plantations (6.30% to 3.10%) and water (5.50% to 5.30%) covered similarly low areas, and the smallest land-cover areas were built-up areas (1.30% to 1.50%) (Figure. 2.2 & Figure. 2.3). The overall landscape changes over the 23-year period appeared relatively low. From 1990 to 2013 plantations had decreased by 50.45%, built-up increased by 19.38%, vegetation increased by 4.07%, water and agriculture remained relatively consistent in area size (Figure. 2.3).

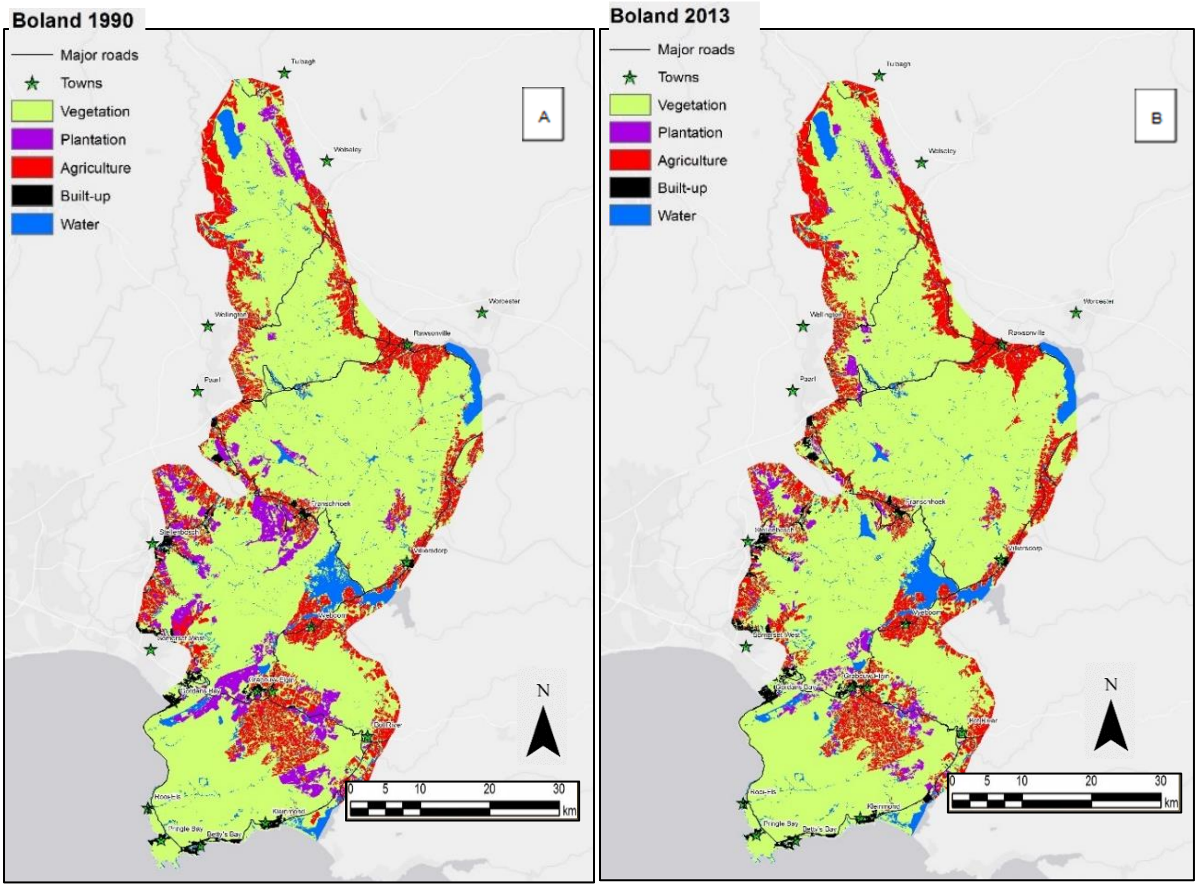


Figure 2.2: The extent of each of the five land-cover classes mapped for the study area A: 1990 and B: 2013. (GEOTERRAIMAGE, 2014).

Shifts between land-covers

In total 3222.66km^2 (88.66%) of area persisted and remained unchanged between 1990 and 2013, while 412.31km^2 (11.34%) shifted to a different land-cover (Table. 2.2). The vegetation land-cover class incurred the largest shift of area from 1990 to 2013 with a gross gain of 245km^2 between 1990 and 2013, due to transformation from plantation (130km^2), agriculture (61km^2), water bodies (48km^2) and built-up (5km^2) land-covers. However, the vegetation land-cover class also decreased by 138km^2 in the same period due to tranformation to agriculture (62.3km^2), water (35.2km^2), plantation (29km^2) and built-up (11.8km^2) land-covers. These various changes in land-cover resulted in a net gain of 107.5km^2 of vegetated land within the BMC (Table 2.2).

Plantations showed more change than persistence from 1990 to 2013, showing a proportionally large decrease (Table 2.2). Of the plantation area, 82km^2 remained and 31.3km^2 was gained from other land-covers from 1990 to 2013, whereas an entire 131km^2 was converted to vegetation and the other 15km^2 was split into the three other land-covers (Table. 2.2).

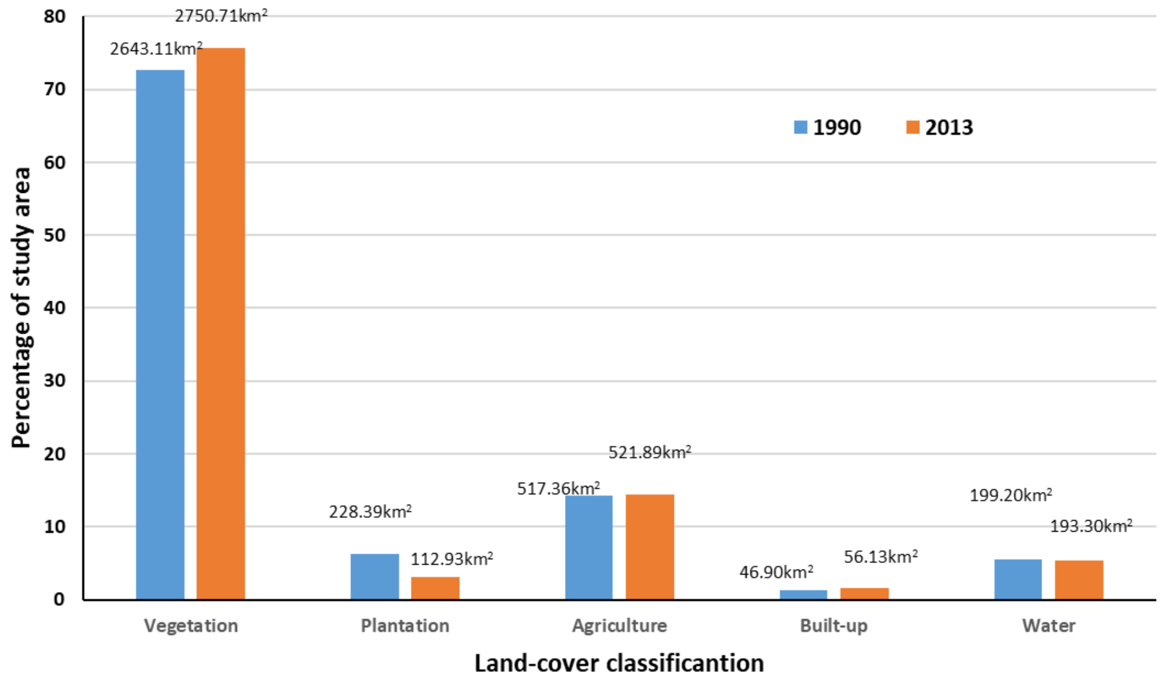


Figure 2.3: Total Boland Mountain Complex land-cover percentages in 1990 and 2013. (GEOTERRAIMAGE, 2014).

The majority of agriculture persisted from 1990 to 2013. The largest loss of agricultural area was a conversion to vegetation (61.1km^2) and a small portion (6.04km^2) to the other land-covers. The few changes to built-up areas were the result of transformation to vegetation. Loss and gain of area cover by the water bodies class were attributed to transformation to and from the vegetation class (Table. 2.2).

Table 2.2: Cross-tabulation matrix of land-cover classes from 1990 and 2013, their totals, gross losses, gains, total changes, net changes and swaps (km^2).

1990	2013									
	Vegetation	Plantation	Agriculture	Built-up	Water	Total 1990	Gross loss	Total change	Net change	Swap
Vegetation	2504.76	29.02	62.30	11.82	35.21	2643.11	138.34	384.29	107.60	276.69
Plantation	130.95	81.63	4.42	1.83	9.57	228.40	146.76	178.06	-115.46	293.52
Agriculture	61.05	1.84	450.23	1.58	2.67	517.36	67.14	138.81	4.53	134.27
Built-up	5.67	0.03	0.40	40.49	0.31	46.90	6.41	22.05	9.23	12.81
Water	48.27	0.41	4.56	0.41	145.55	199.20	53.66	101.42	-5.90	107.32
Total 2013	2750.71	112.93	521.89	56.13	193.30	3634.97				
Gross gain	245.94	31.30	71.67	15.64	47.76					

Overall 1.2km^2 of vegetation in 1990 was transformed to agriculture by 2013, 6.2km^2 of vegetation was built over by 2013 and plantations were replaced by more vegetation than what they replaced. Water bodies dried out partially and exposed areas for vegetation to grow. This gain in area from plantations converted to vegetation by 2013 was greater than the area of vegetation that was lost to all four other land-covers by 2013 (Figure. 2.4).

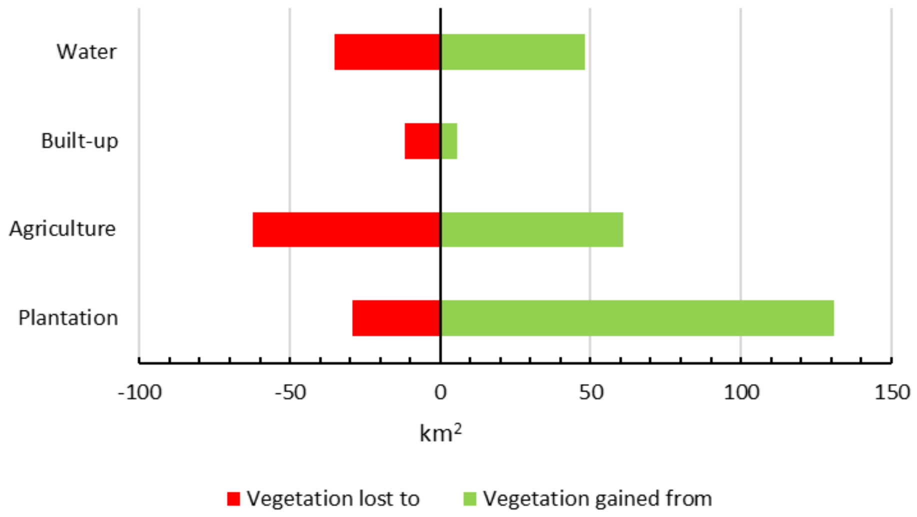


Figure 2.4: Area (km^2) of vegetation gained and lost to other land-uses from 1990 to 2013 in the study area

These land-cover changes to vegetation spanned across many visibly small patches and a few visibly larger areas (Figure. 2.5A). Gain of vegetation from plantation (Figure. 2.5C) was visable south east of Franschhoek, north of Kleinmond, east of Grabouw, between Grabouw and Botriver, south of Tulbagh, along the Stellenbosch Mountains and in the south east cornor of the West Hawequas MZ. Fewer large patches of vegetation were lost to plantation development east of Paarl and Wellington, north of Grabouw, on Simonsberg MZ and south of Tulbagh. Other large patches of gain to vegetation were from agriculture (Figure. 2.5B) near Somerset West, northern West Hawequas MZ, west of Grabouw, east of Kleinmond (Figure. 2.5E) along the north of the Theewaterskloof Dam. Loss of area in the vegetation class were due to transformation of larger patches to agriculture south of Tulbagh, water east of Franschhoek and from built-up land-covers south east of Paarl, Gordans Bay and between Kleinmond and Botriver (Figure. 2.5B&D).

The CWBR had the same proportion of land-cover and changes to each class as throughout the BMC. This included a large gain of vegetation that mostly replaced plantations and agriculture, and a few square kilometres of built-up land (Figure. 2.7). Approximately 92.11km^2 of vegetation was lost and converted into 40.0km^2 of agriculture, 27.0km^2 of water, 21.0km^2 of plantations and a small >5.0km^2 of built-up land-covers.

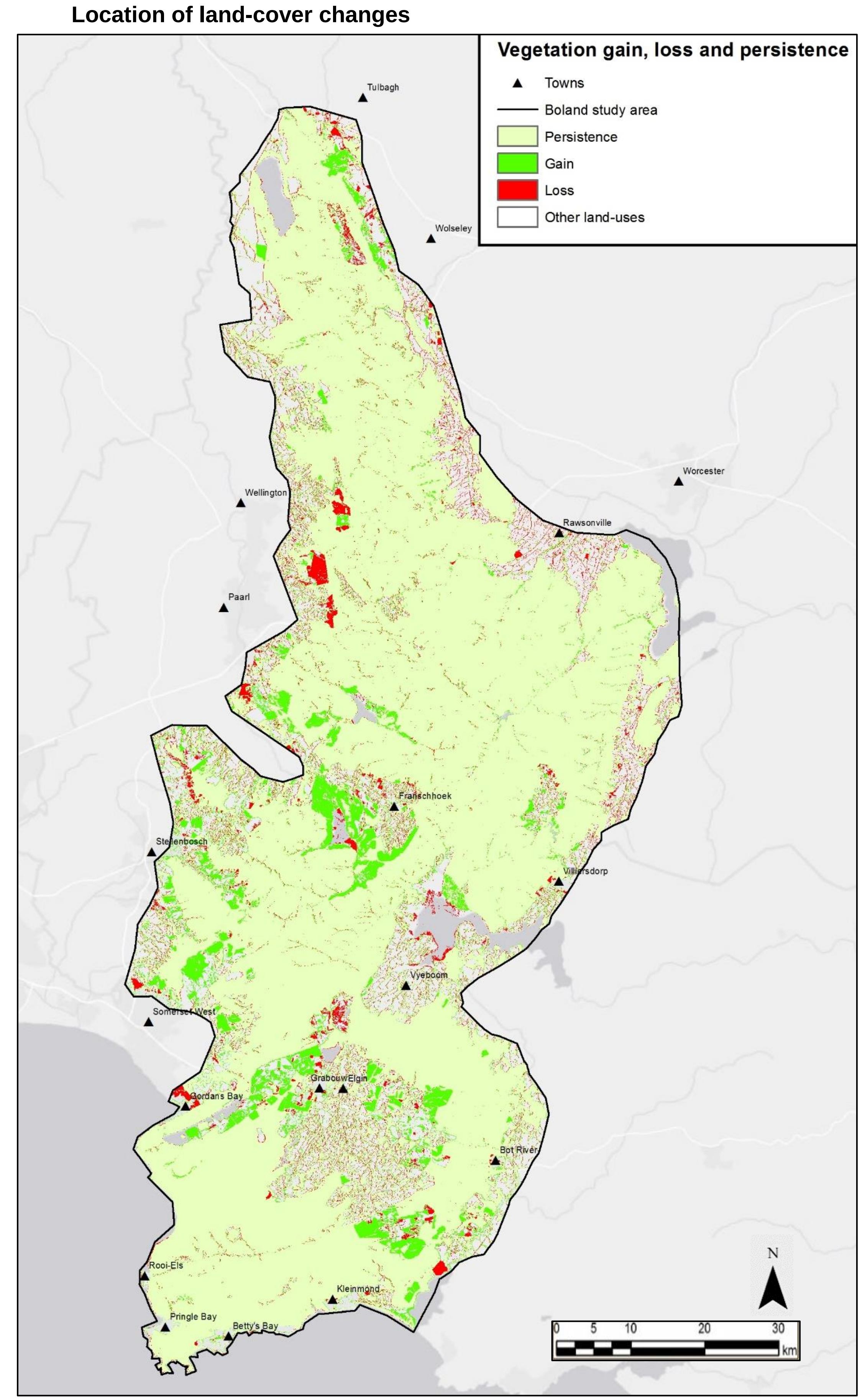


Figure 2.5A: Loss, gain and persistence of vegetation land-cover class across the study area.

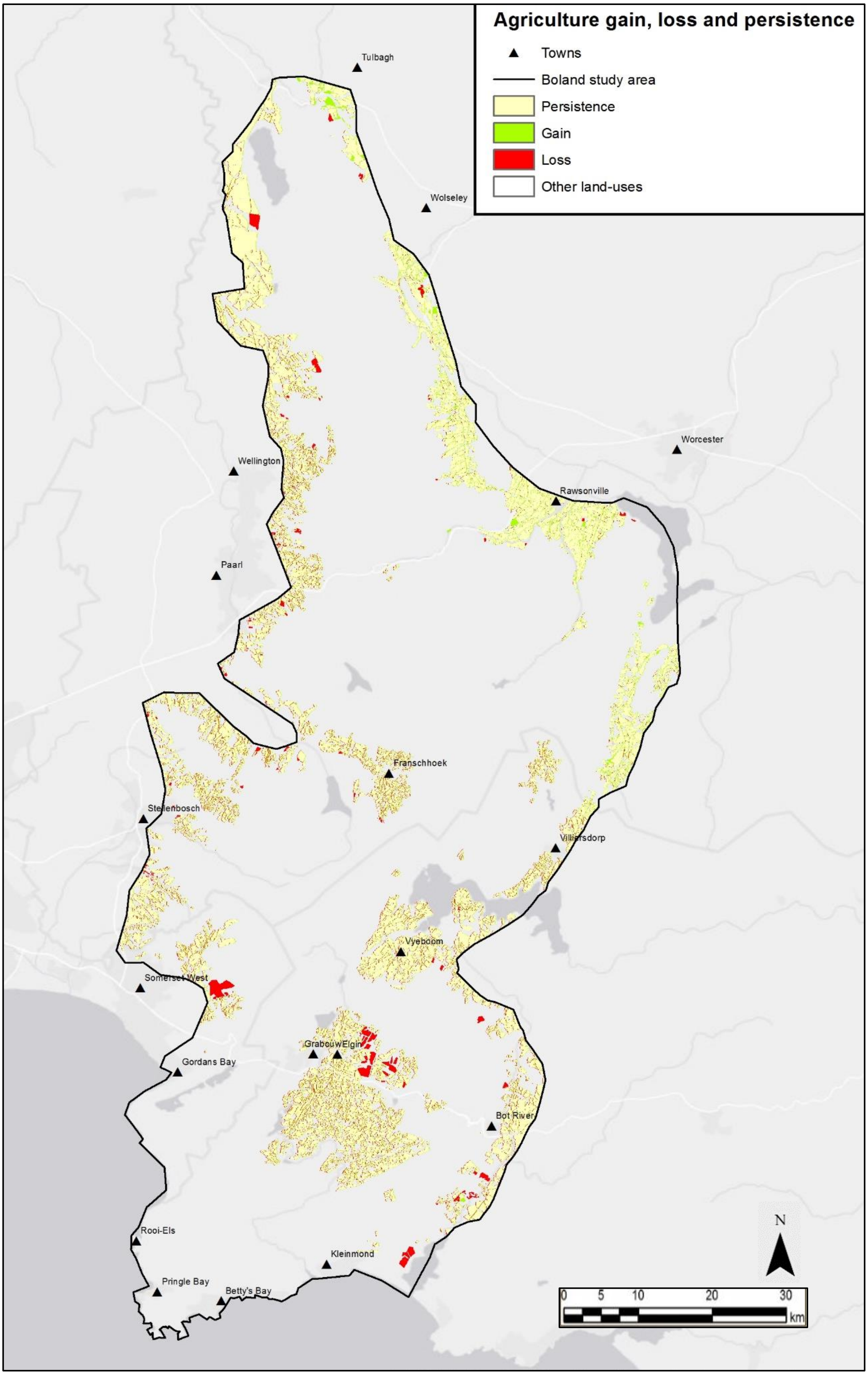


Figure 2.5B: Loss, gain and persistence of agriculture land-cover class across the study area.

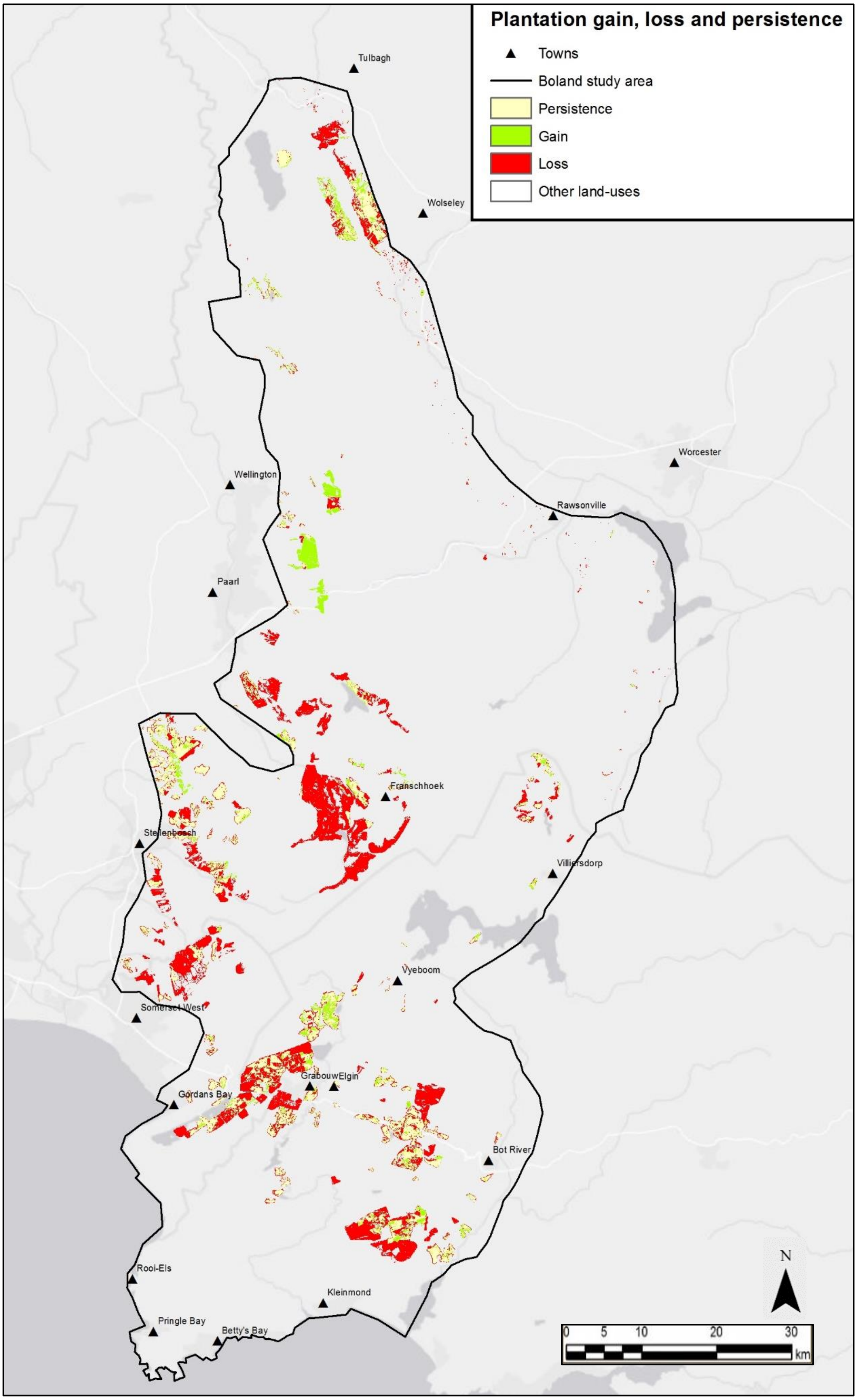


Figure 2.5C: Loss, gain and persistence of plantation land-cover class across the study area.

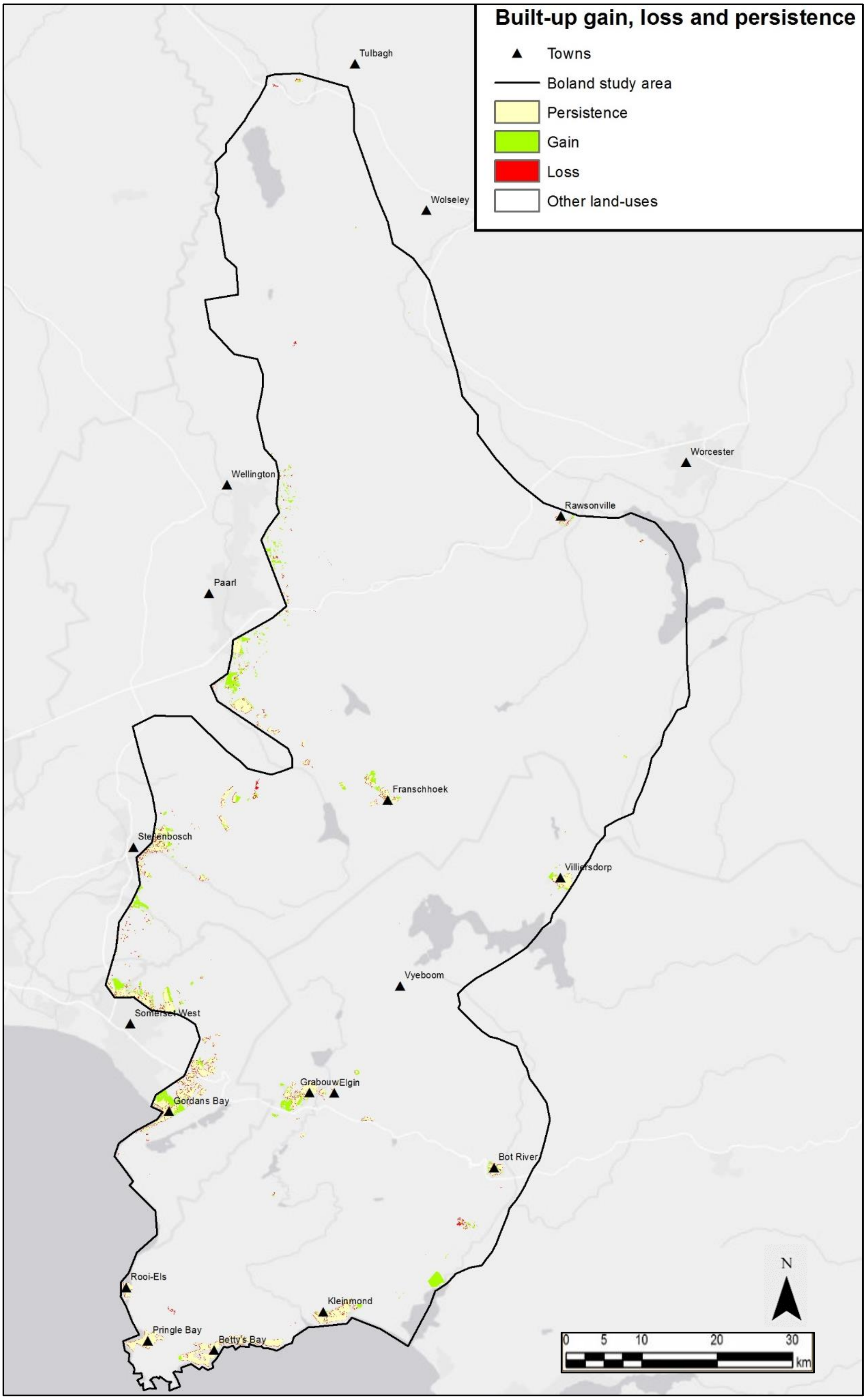


Figure 2.5D: Loss, gain and persistence of built-up land-cover class across the study area.

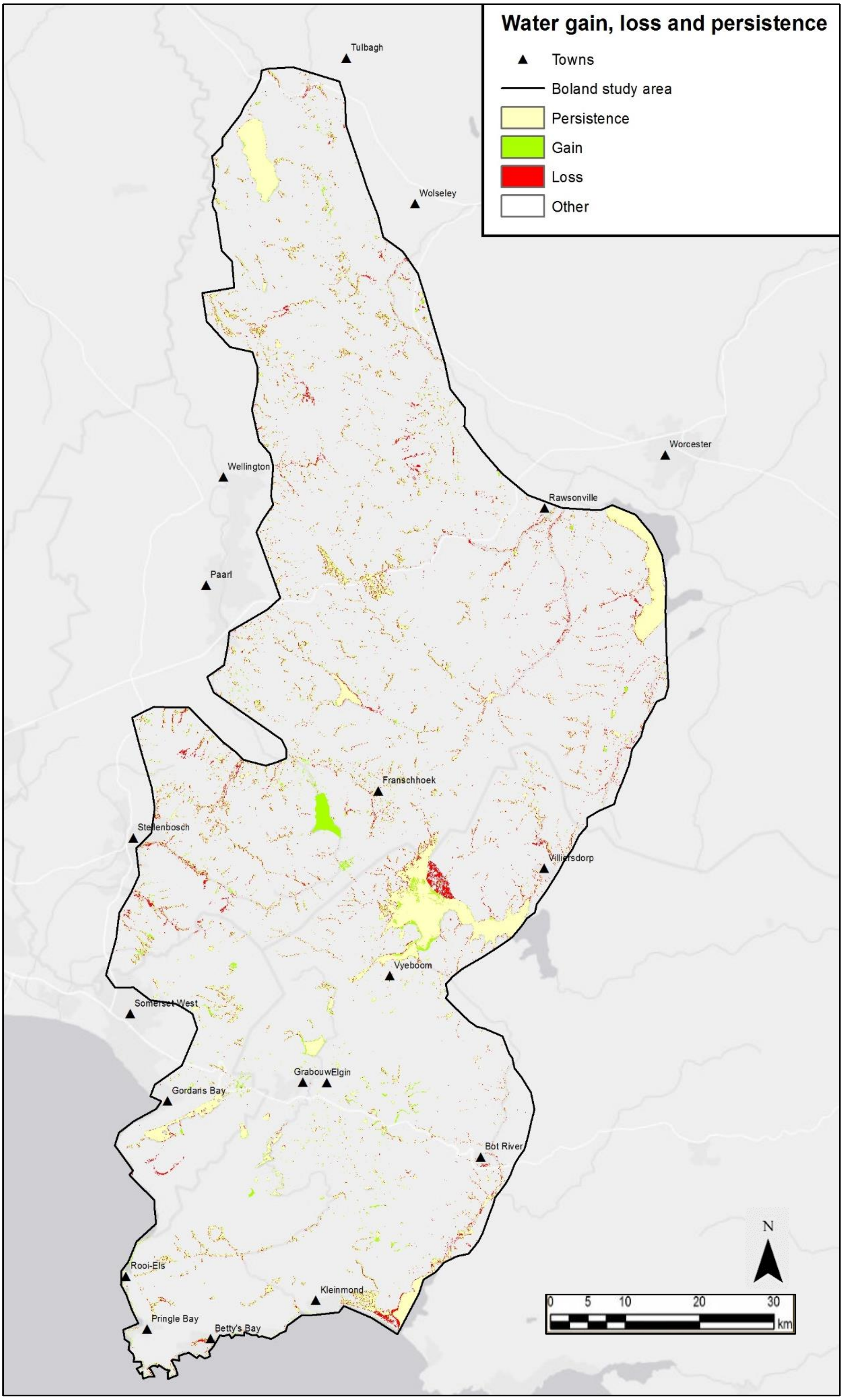


Figure 2.5E: Loss, gain and persistence of water land-cover class across the study area.

These proportions existed similarly in the KBC, with a much greater decrease in plantations from 1990 (10.0%) to 2013 (5.2%) and a slight decrease in agriculture. The loss of these land-covers was mostly converted into vegetation (45.8km^2) (Figure. 2.8). The patterns of gains and losses to vegetation in the KBC were similar to the CWBR. However, the KBC had a larger percentage of built-up land-covers, which replaced vegetation, when compared to the CWBR.

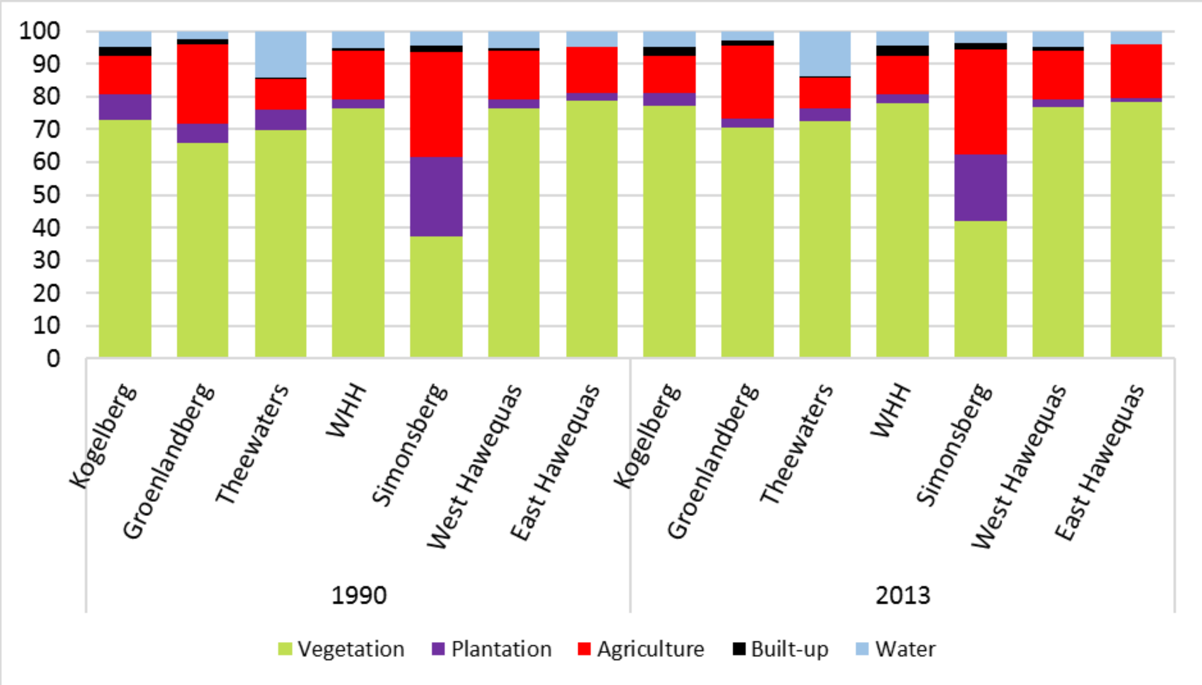


Figure 2.6: Percentage (y-axis) of mountain zones made up by land-cover classes in 1990 and 2013

Simonsberg MZ was a visible outlier, showing substantial differences from the rest of the study area, as the only mountain zone with less than half its area made up of the vegetation land-cover in both 1990 (37%) and 2013 (42%) (Figure. 2.6). However, the vegetation class did increase between the years. Simonsberg MZ had the most agriculture porportionally when compared to the other mountain zones, for both time periods. Plantations were the highest contributors of vegetation loss, replacing a total of 8km^2 (Figure. 2.6). The East Hawequas MZ was the only MZ to indicate an overall increase in agriculture by 2013, that resulted in the only decrease in vegetation, as it expanded into 21.5km^2 of the latter (Figure. 2.6). The West Hottentots-Holland MZ showed the greatest decrease in plantations that were replaced by 47km^2 of vegetation and the built-up land-covers increased to over three-times what they covered in 1990 (Figure. 2.6).

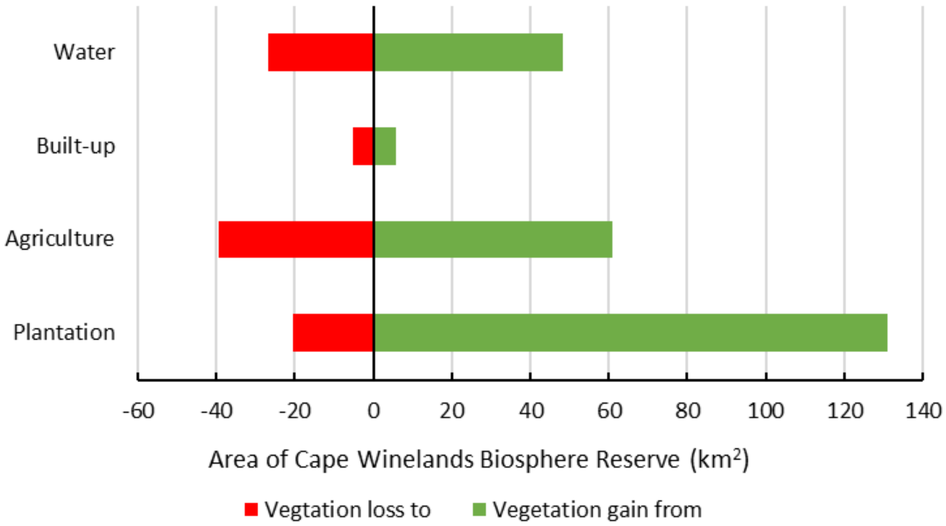


Figure 2.7: Area (km^2) of vegetation gained and lost to other land-covers from 1990 to 2013 in the Cape Winelands Biosphere Reserve

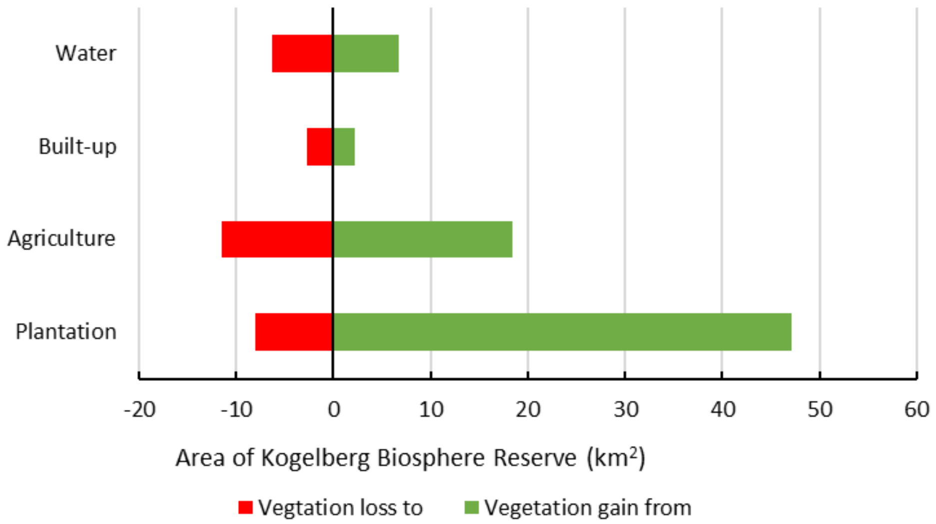


Figure 2.8: Area (km^2) of vegetation gained and lost to other land-covers from 1990 to 2013 in the Kogelberg Biosphere Reserve.

2.4.2 Wildfire Frequency and Spatial Patterns

Number of fires burnt

Since 1957, 1193 wildfires burnt in the BMC (CapeNature, 2017). These covered an overlapping total of 8033.48km^2 of land-surface. Many of the ignition sources of these fires could not be determined, however 211 were managed fire operations (block burning, fire break creation and refuse burning) and 113 were mechanical accidents (machinery, power lines, vehicles and trains, dynamite and other). Humans started 352 known fires through arson, smoking, cooking, for warmth, while smoking beehives, farming and by unattended children, and only 66 were natural ignitions (lightning strikes, rock falls and other) (CapeNature, 2017) (Table 2.3).

Table 2.3: Percentage of fires from each ignition source per 15-year period. (CapeNature, 2017).

Period	Percentage of fires from each ignition source				
	Natural	Fire operation	Direct human cause	Mechanical	Other/ Unknown
A 1957-1972	3.77	26.42	7.55	0	62.26
B 1973-1987	5.93	28.85	18.97	3.95	42.29
C 1987-2002	6.75	28.57	27.78	6.35	30.56
D 2002-2017	5.15	7.02	36.04	13.57	38.22

The number of fires that burnt per 15-year period differed significantly from 1957 through to 2017 ($F_{(3, 28)} = 48.015$, $p<0.001$) (Figure. 2.11). The average number of fires that burnt per year during period A were significantly less than those which burnt in the following periods B and C ($p<0.001$). Periods B and C did not differ significantly from each other. The average number of fires for the most recent period D, was significantly higher than any of the three previous periods ($p <0.001$) (Figure. 2.11).

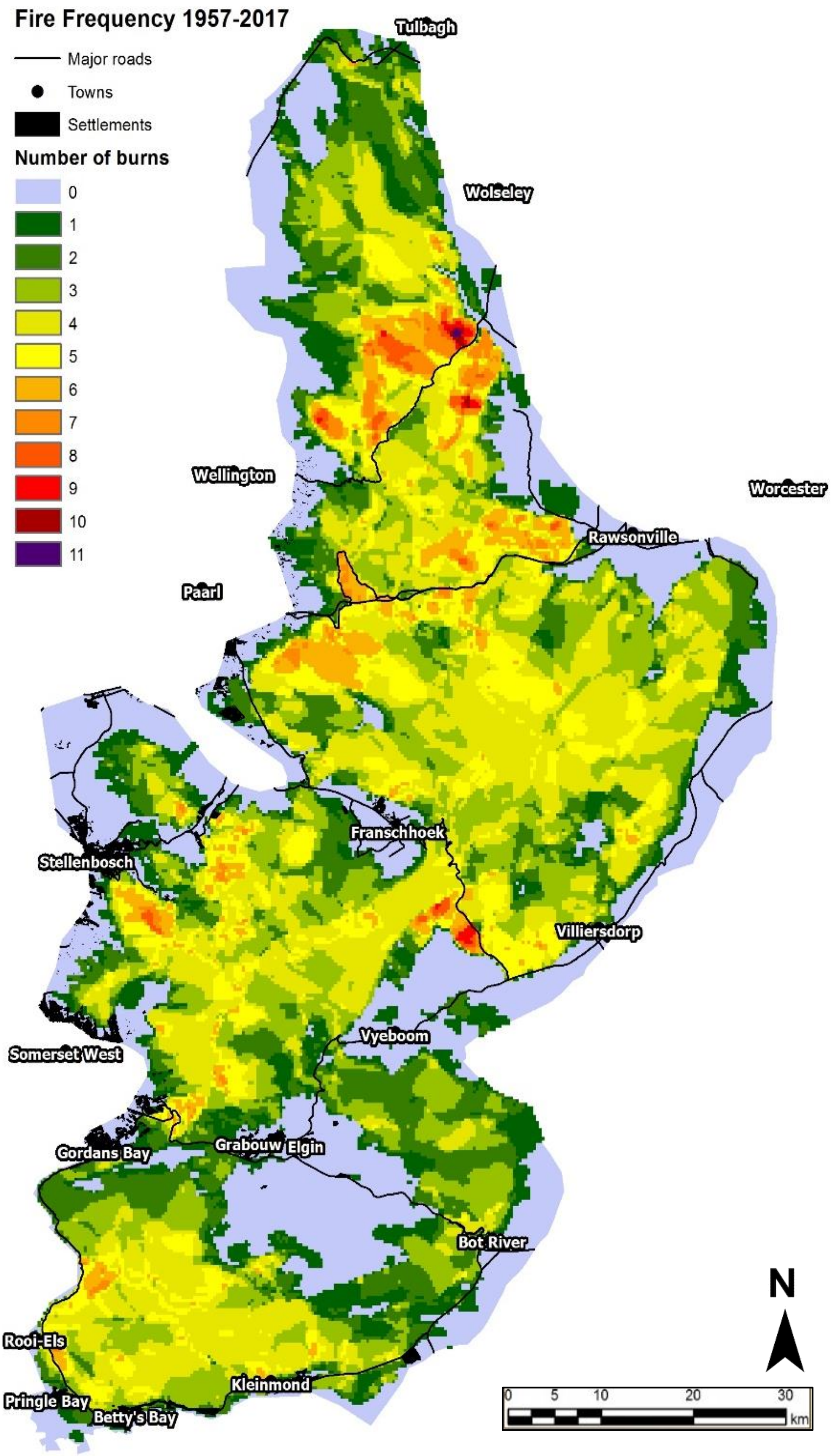


Figure 2.9: Fire frequency regime from 1957 to 2017 of entire study area (CapeNature, 2017).

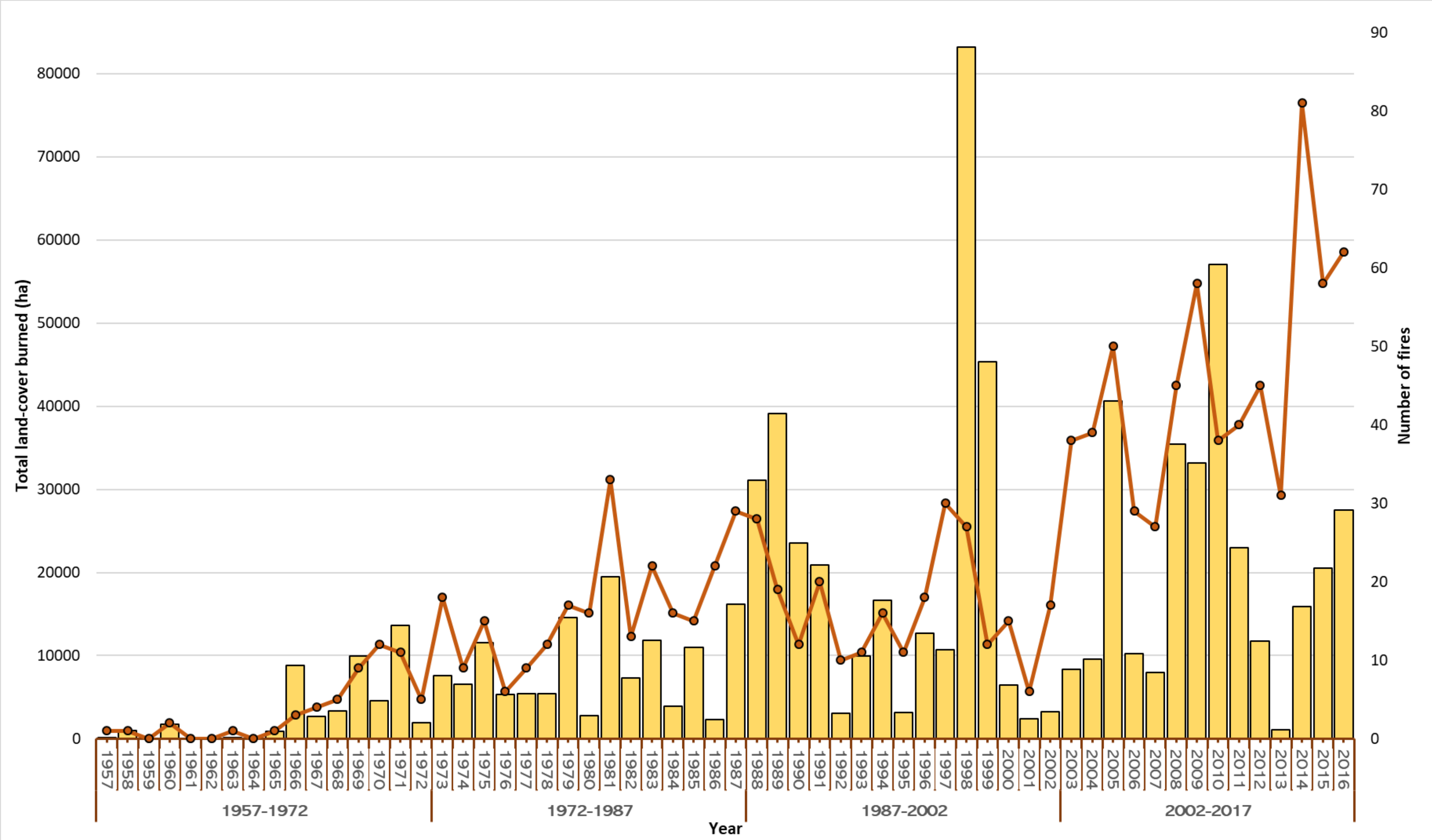


Figure 2.10: Bars of total land cover (ha) burnt (left y-axis) per year and line illustrating fires per year (right y-axis) from 1957 to 2017 (CapeNature, 2017).

Area burnt

A significant increase in the total land-cover that burnt per year across periods was seen ($F_{(3, 28)}$ =10.533, p<0.001) (Figure. 2.13). During period A, fires disturbed significantly less land-cover than any period in the following 45-years, from period A to B (p=0.011), to period C (p=0.034) and to period D (p=0.003). Periods C and D both had burnt areas with sizes not significantly different to one another (p=0.999) (Figure. 2.12). The size of individual fires per year showed a weak, negative correlation to the mean area burnt by fires over time (r_S = -0.380, p<0.001).

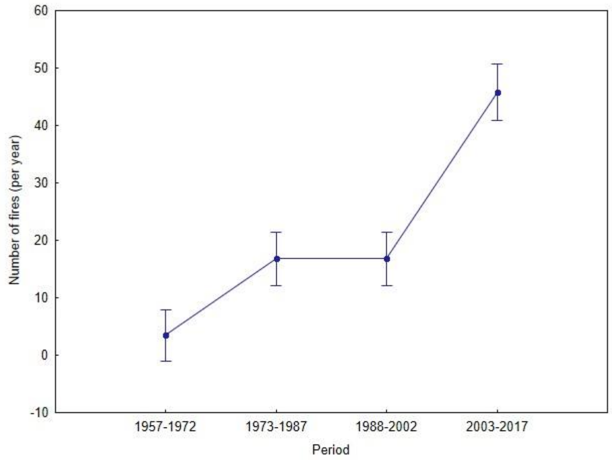


Figure 2.11: Average number of fires that burnt per year for each historic period across the study area. Vertical bars denote 0.95 confidence interval

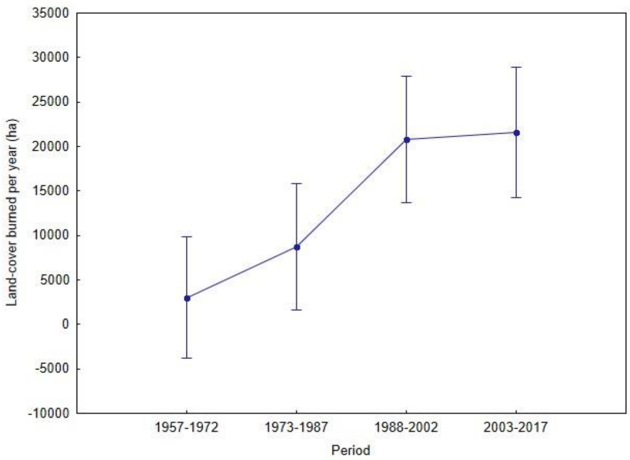


Figure 2.12: Average area of land-cover burnt per year for each historical period across the study area. Vertical bars denote 0.95 confidence interval

Fire Return Intervals (FRI's) and variance in fire locations

The FRI's for period A varied between the two biosphere reserves and the MZs. The CWBR required the highest number of years to burn, approximately 140-years, while the KBC required a shorter period of 74-years. The number of years that it took for the whole study area to burn, decreased during period D to very close average FRI's for all MZs and both biosphere reserves, and reached a period of about 23-years (Figure. 2.14).

Over the course of period A, there were great differences in the FRI's of each MZ. The Kogelberg MZ (57-years), the West Hawequas MZ (104-years) and East Hawequas MZ (97-years) had the shortest FRI's, while Theewaterskloof Basin MZ (1494-years), West Hottentots-Holland MZ (1021-years) and Simonsberg MZ (716-years) had the longest FRI's. From 1957 to 2017 a significant variation in the rank-distributions of each MZ's FRI between each period was displayed by the Friedman's test (X^2=17, df=1, p=0.001).

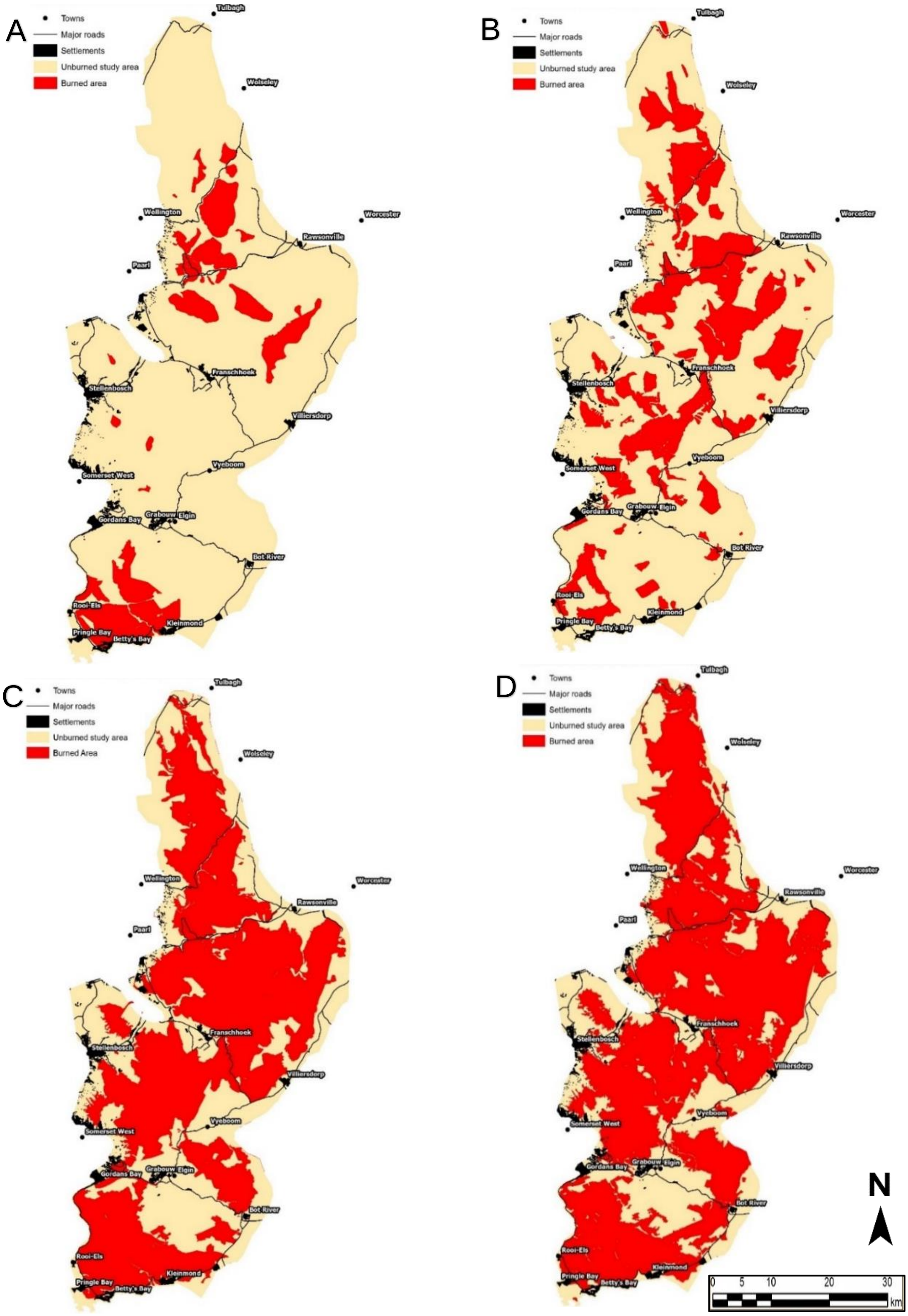


Figure 2.13: Area of study area burnt per period of 15 years. (A: 1957 to 1972. B: 1972 to 1987, C: 1987 to 2002 and D: 2002 to 2017) (CapeNature, 2017).

A Wilcoxon signed-rank post-hoc test with a Bonferroni adjustment applied, among the individual periods (A, B, C and D) (p=0.008, -1.572>Z>-2.366, p>0.17) confirmed there was a significant change in the average FRI's from 1957 to 2017 (Figure. 2.14). For all mountain regions, the FRI shortened with time and during period C (29-years) and period D (23-years) little variance was exhibited in the FRI's of each mountain zone. The East and West Hawequas MZs continuously had the shortest FRI, whilst Groenlandberg and Simonsberg MZs consistently had the longest FRI (Figure. 2.14).

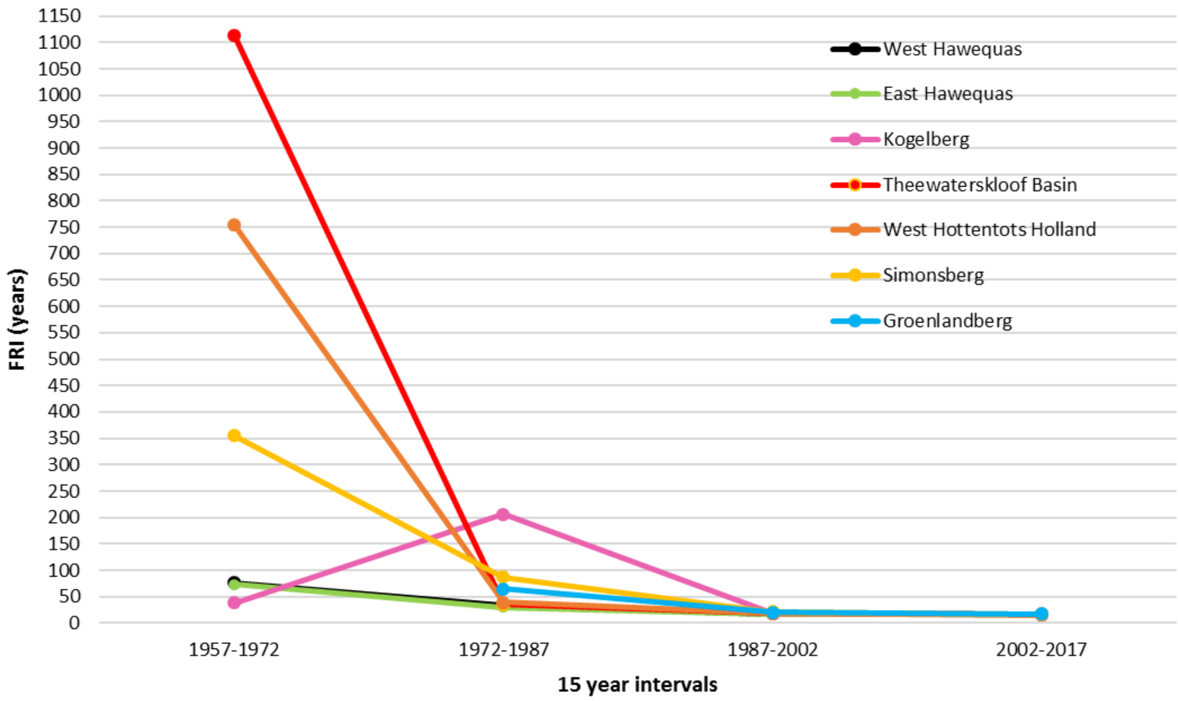


Figure 2.14: Mean Fire Return Intervals (FRI) since 1957 to 2017 over this study's four 15-year periods for each mountain zone

Fire frequency showed an inverse relationship to the distance from settlement edges. A location with a higher fire frequency was therefore located at greater distances from settlements edges (r =0.137, p<0.001). No relationship was determined between fire frequency and distance from roads (r=0.083, p>0.05).

2.5 Discussion and conclusion

2.5.1 Land-cover change

Vegetation

From 1990 to 2013, the vegetation class area in the BMC study region increased by a total of 107.5km^2. This shift in land-covers is beneficial to medium-sized mammals in terms of it representing an increase in habitat. Within the study area, the vegetation class comprised of the main proportion of land-cover, followed by agriculture, forestry, built-up and water land-cover classes for both study years. The dominance of the vegetation class was expected as the study area consisted of over 60% PA's as the biosphere reserves' core zones. Few quantitative changes and a consistent structure over the 23-years indicates land-cover allocations do adhere to the zones of UNESCO Biosphere Reserves (Pool-Stanvliet, 2013; Pool-Stanvliet & Giliomee, 2013). As von Hase et al. (2010) stated, most of the lowland vegetation in the Cape, which was of greatest use to agriculture, had already been cleared in the 1960's, and it was unlikely for more to be cleared now. However, a gain in vegetation was a much more beneficial outcome for medium-sized mammals than expected during this study period.

The gain in vegetation land-cover exceeds Boshoff et al. (2001) roughly estimated minimum home range of an individual female leopard in the Fynbos biome (Boshoff et al., 2002). This may imply that this increase in the vegetation class is ecologically sufficient to benefit the majority of the study species, with home ranges generally smaller than the 107.5km^2 (Boshoff et al., 2001). However, that said, female leopard home ranges in the Boland region average approximately 80km^2, which equates to the home range of 1.5 female leopards. Additionally, bearing in mind that this area is not continuous, but rather patchy and represents only 3% of the total study area. The decrease in plantations, agriculture and built-up land-covers and conversion to vegetation allow for a landscape made up of a more heterogeneous land-cover class, with fewer anthropogenic threats to mammals (Smith et al., 2004). Overall the gain in vegetation may not significantly benefit medium-sized mammals. However, this small increase with the low loss of habitat to land-cover change indicates that the way in which human development occurred over the 23-year period was a lower threat to mammals than hypothesized.

Agriculture

Shifts of vegetation to agriculture generally appeared within established agricultural patches and along edges between farms. Agricultural expansion was thus less intrusive into buffer habitats adjacent to the core than expected. The core and protected areas in the BMC consist of mountains and are generally sloped, therefore this study did not show agriculture moved up into typically higher slopes between 1990 and 2013. Thus, the buffer zones in the study area do not show obvious inclinations of viniculture moving to steeper slopes as Hannah et al. (2013) predicted will happen between 2000 and 2050. Reasons for this may be the physical restrictions of these steep, shallow-soiled, mountain slopes that are unsuitable for most agriculture (Hannah et al. 2013; Rouget et al., 2003). The current study showed no evidence by 2013, that Fairbanks et al. (2004)'s predictions of key habitat becoming viniculture, was a concern. CapeNature's enforcement of development regulations as well as their and WWF's conservation stewardship agreements may be accredited to the limited advancement of agricultural encroachment into buffer vegetation (Duffell-Canham et al., 2017; Hannah et al., 2013; Pool-Stanvliet et al., 2017; von Hase et al., 2010). Another factor may be that many landowners value maintaining areas of natural habitat (Pool-Stanvliet et al., 2017). Advances in production technology as well as environmental restrictions (drought and climate change) may too have aided in agricultural intensification rather than expansion since 1990 (University of Stellenbosch Business School, 2018).

The only area where there were larger shifts from vegetation to agriculture compared to other MZs was on the edge of the East Hawequas MZ (near Tulbagh, Figure. 2.5B). This vegetation loss to agriculture was located close to the corridor leading out into the Groot Winterhoek Mountains that form an extended mammalian range (Wilkinson, personal communication, 2018). As large-bodied, long-distance dispersers, leopards depend on functional connectivity between natural habitats (Fattebert et al., 2013). At the edge of the study area, this corridor consists of a 1280m breadth of natural vegetation between agriculture on either side. Beier (1995) has previously reported more narrow corridors had lower success rates of cougars crossing them. Henschel et al. (2011) concluded leopards can persist with low prey availability at low densities, but it is access to wider habitat ranges that allow for this survival. This is the only visible vegetated corridor that is uninterrupted by agriculture, from the BMC to the northern PA's. Although leopards can disperse over agriculture, continued vegetation enhances this movement as well as the likelihood of other medium-sized mammals accessing more habitat (Beier, 1995; Harris & Scheck, 1991).

The R46 is a main regional road that winds through this narrowed section of natural vegetation. The Western Cape Government's Road Network Information System shows that by 2015 sections of that road had five times more vehicles using it daily than in 1989 (https://rnis.westerncape.gov.za/rnis/rnis_web_reports.main#null). The road's role as a deterrent and barrier to some species, particularly during high traffic hours, would likely have increased since 1990, as well (Klar et al., 2009). Narrow corridors with major roads increase the risk of mortality from vehicle-wildlife collisions, particularly to dispersing individuals that are forced to cross these roads (Barthelmess, 2014; Beier, 1995).

Simonsberg MZ's small size and isolation by agriculture and partially by built-up land-covers (including roads) likely enhance the edge effects it experiences (Newmark, 2008; Parks & Harcourt, 2002). The effects of having a large outer edge to core surface area was visible in the current study, as the MZ showed different compositions and changes in land-clover classes (Newmark, 2008; Pardini et al., 2018). This MZ still maintains access to individual leopards, despite no visible, uninterrupted vegetation corridors to other PA's. Crooks (2002) showed that isolation and the small size of a habitat effected mountain lions, bobcats and coyotes, which are territorial, dispersing predators, like leopards. Corridors and the preservation of all sized PA's therefore can aid in both leopard and other medium-sized mammals habitat availability and accessibility (Fattebert et al., 2013).

Plantations

Felling and the decommissioning of plantations was the largest contributor of any land-cover shift to an increase in the vegetation class area (130.95km^2). This made many large-sized patches in various locations available for vegetation rehabilitation. Some shifts from plantations to vegetation were expected, however not on this large-scale (Fairbanks et al., 2004; Ruiz, 2003). Rebelo et al. (2019) found rehabilitated fynbos supports small-sized mammal communities better than plantations. This would presumably have a bottom-up affect and allow for an increase in medium-sized mammal species richness and abundance (Iezzi et al., 2018). Rehabilitation of plantations is key to sufficient recovery of fynbos ecosystems and returned functionality for medium-sized mammals (Holmes & Richardson, 1999). Both passive and active restoration methods were used on pine plantations in the BMC during this land-cover's decommission (Louw, 2010; Van Wilgen, 2015). The intensity of rehabilitation methods depends on various factors (such as the age of felled pines), and both methods require follow-up control plans (Galloway et al., 2017; Hitchcock et al., 2012). Large areas of pine were replaced by vegetation from 1990 to 2013, but the level of successful support to fauna biodiversity, as seen in Rebelo et al.'s (2019) study, is dependent on rehabilitation management (Holmes & Richardson, 1999).

This widespread felling of pine plantations was due to planned clearing in the Western Cape since 1999 by the South African Forestry Company (SAFCOL) (Ruiz, 2003; Tizora et al., 2016). Land-cover changes are therefore moving in the direction that Ruiz (2003) suggested: that fynbos is the most suitable land-cover to replace these plantations. We expect some strong contrasts between pine plantations and both new and established areas of vegetation, because of Armstrong et al.'s (1996) description of plantations being so desolate of vertebrate diversity. Pine plantations have less severe effects on larger sized mammals and are able to benefit some generalist species, but their homogenous vegetation structure and lack of understory coverage make them poor sources of shelter and nutrition (Armstrong et al., 1996; Golley et al., 1975; Iezzi et al., 2018; Rebelo et al., 2019). These factors would make them barriers to some mammals, therefore the large areas felled from 1990 to 2013 would have increased the size of habitat available and eased the accessibility to more of the BMC for medium-sized mammals (Brodie et al., 2014; Iezzi et al., 2018; Lindenmayer & Hobbs, 2004;

Merriam, 1991). New vegetation patches replaced plantations between the Kogelberg, Theewaterskloof Basin and West Hottentots-Holland MZs, thus better connecting the three MZ's vegetation land-covers.

The decrease in plantations lowers the risks of altered fire regimes (Mostert et al., 2017). Pine plantations burn at high intensities for longer periods and have severe impacts that lower faunal survival (Davis & Burrows, 1994; Duncan & Schmalzer, 2004; Mostert et al., 2017). Another ease on the ecosystem from the removal of pine plantations is the fewer opportunities of spread and germination of seeds (van Wilgen, 2015). The pine trees are highly invasive and able to alter natural habitats in the above-mentioned ways without human-driven physical clearing of vegetation and propagation of pines (Richardson, 1998; Richardson et al., 1994).

Water

Little can be discussed about changes to the water landcover. Changes in water surface area are always recorded over time in land-cover analyses (Sonestrom et al., 2009). This is because natural water cycles fluctuate, and the class includes both dams and natural water bodies (Sonestrom et al., 2009). The Berg River Dam was one of the noticable developments that contributed 4.88km^2 to the water class in 2008 (Ruiz, 2003). The dam was built with consideration for the environment and did not replace natural vegetation, only plantations (Ruiz, 2003). The surrounding area consisted of plantations that were converted to vegetation by 2013 and are intended for fynbos conservation (Ruiz, 2003). The decrease in water surface area could be the result of the major drought experienced during the study period, but that cannot be ascertained from this study.

Built-up

Changes to built-up areas did not strongly reflect Tizora et al.'s (2016) finding that development was expanding significantly in the Cape Winelands and was only seen around Paarl. Their study's recorded changes had likely moved into other vegetation areas of the Cape Winelands, away from the study area. However, 11km^2 of newly built-up land-cover that replaced vegetation is quite a large area of development. These built-up land-covers included; industrial buildings (greenhouses, sawmills, brick factories and sand mines), commercial shopping markets and residential land-uses that took up the majority of increased built-up areas (including formal housing estates and golf courses, suburban housing, government subsidy housing and informal settlements). From 2013 to 2017, many of these built-up land-uses had expanded visibly further into more preciously vegetated areas, according to satellite images (Google, 2019). The majority of major towns had some form of residential built-up area increase, which reflects the known increase in the human population size since 1990 (Statistics South Africa, 2018). These housing types ranged from high-income housing estates to low-income, subsidy houses and informal settlements. Built-up land-covers generally make more rigid barriers to medium-sized mammalian movement and utilisation (Sauvajot et al., 1998). They also can emit a more severe and a greater range of anthropogenic effects into natural ecosystems (Sauvajot et al., 1998).

Housing estates generally take up large expanses in the BMC, with low densities of humans, large areas of suburban vegetation and high security fences. These and suburban houses may allow some species to utilise or move through the land-covers because of their vegetation and spaces between buildings (Racey & Euler, 1982). High security fences may, however act as barriers to many terrestrial mammals and may isolate habitats from an ecosystem without changing the land-cover class (Newmark, 2008). This is a stark contrast to typical subsidy housing and informal settlements, which have many structures clustered closely to one-another, high human densities per structure, limited space between houses, little vegetation and often lack distinct parameters (Govender et al., 2011). Informal settlements are much stronger barriers with unlikely penetrability for medium-sized mammals. The high densities of humans, who are often impoverished, are what may cause wider ranging negative impacts on faunal communities in the BMC. The unclear and unmonitored parameters of informal settlements increase the inhabitant's ability to spread into surrounding landscapes and ecosystems (Sauvajot et al., 1998).

The hunting and consumption of bushmeat near PA's is often driven by food insecurity, due to poverty (Rentsch & Damon, 2013). High incidences of illegal hunting of several local medium-sized mammals were reported in the area by Nieman et al. (2019). Other impacts associated with higher human population densities include; the introduction of free-roaming domestic animals (MacDonald et al., 1998), increased fire ignition sources (Sauvajot, 1995), pesticides and toxins (Serieys et al., 2019), human-wildlife conflict (Martins & Martins, 2006) and increased traffic, with more developed road networks that act as greater movement barriers and mortality risks from wildlife-vehicle collisions (Klar et al., 2009; Newmark, 2008).

2.5.2 Fire Regime Changes

Multiple elements of fire regimes displayed significant changes over the 60-year period in the study area. The increasing number of fires as time progressed with the average size of individual fires not increasing, indicated that many small fires caused more area to burn over shorter intervals (Archibald, 2016). These results are similar to those of other studies in nearby sections of the Fynbos biome which found an increased area burning per year in recent periods (Forsyth & Van Wilgen, 2008; Kraaij et al., 2013). Fire frequency also showed significant increases throughout the BMC, as Gumbi (2011) found in the Kogelberg Nature Reserve. However, it is unlikely that climate change is the primary driver of increased fire frequencies, as the above-mentioned study concluded. This would have been visible as an increased size of individual fires over time (Archibald, 2016). Results from the study support that of Southey (2009), that increased human-caused ignition sources are the primary driver of the increased fire frequency in the BMC. Southey (2009) reported patterns of an increasing number of ignitions in the Hottentots-Holland Nature Reserve, and this is supported by the data for the rest of the BMC. Although many of the ignition sources were unrecorded, the majority that were known were directly and indirectly caused by human activities. (Archibald, 2016; Kraaij & van Wilgen, 2014). Human population increases and advances in and more widespread use of machinery, are some likely causes of more ignitions in recent years (Chan et al., 2011; Gumbi, 2011; Kraaij et al., 2013; Mack & D’Antonio, 1998; Southey, 2009). Poor fire control and management methods, human-ignited fires (via arson) and accidents were often key drivers for the fire regime changes detected in this study (Chan et al., 2011; Mack & D’Antonio, 1998; Syphard et al., 2009; van Wilgen et al., 2010).

In contrast to many human ignition sources, this study found that patches of land with higher fire frequencies over the entire study period were located further from human development edges. This was unexpected as many fire ignitions are associated with near distances to human settlements (Forsyth & van Wilgen, 2008). It was however, a similar trend to that of Archibald et al. (2009) who determined that human population density has a negative correlation to burn areas. Globally, studies have found both similar and the inverse results to the current study, which have been attributed to the general management strategies and responses to fires in local towns (Duncan & Schmalzer, 2004; Elliott et al., 2009; Wells et al., 2004). It seems probable that a higher frequency is influenced by the difficulty for management to access these remote locations in order to control fires. No relationship was however found between distance from major roads and fire frequency. A greater urgency to extinguish fires closer to settlements is therefore the more likely cause for increased fire frequencies rather than accessibility in the study area (van Wilgen et al., 2012).

FRI’s generally always differ from one period to another (Van Wilgen, 2013). However, the constant interval decrease since 1957 in the BMC is alarming to all aspects of the ecosystem (Forsyth & Van Wilgen, 2008). This conversion from various MZs burning with a range of FRI’s to approximately the same FRI for the whole study area defies the natural heterogeneity in fire regimes that van Wilgen (2013) noted (in Radford, 2008). This increase in both the frequencies of fires and total area burning across the BMC, since the 1950's are similar to the fire regime changes that Gumbi (2011) documented in the Kogelberg Nature Reserve, until 2006. The current study then indicated that these similar fire regime patterns continued into 2017. It seems probable that without a significant intervention or major disturbance, the fire frequencies will continue to increase. Fire management did go through a decline of resources and input during the 1980's, which may too have driven the significant fire regime changes from the second half of the study period (Kraaij & van Wilgen, 2014; van Wilgen et al., 2012). The illustrated fire regime in this study matches the homogenous pattern that the majority of the world's fire regimes are conforming to, because of human influence (Archibald, 2016).

The average FRI of 23-years for the BMC in 2017, should allow for mammalian fauna to persist. Kraaij et al. (2012) noted that fauna need an ecosystem with variable fire frequencies, where some sections of vegetation have been established for 15 to 20-years. The continued trend of increasing fire frequencies, consistently across the landscape is however concerning in terms of the potential impacts on future mammal populations. A lower heterogeneity in the fire ecology of the BMC landscape may disrupt mammal diversity in the future (Andersen et al., 2005; Bradstock, 2008). Homogenous fire regimes also limit the available, unburnt habitat as refuges for individual mammals' survival (Andersen et al., 2005; Bradstock, 2008; van Wilgen et al., 2010). The large areas of shifts by land-covers into vegetation (in areas like the West Hottentots-Holland, East Hawequas and Kogelberg MZs), as habitat, likely provided some relief to medium-sized mammals from the increased number of fires burning (Figure. 2.5A). Many species are likely able to also use the buffer land-covers such as agriculture and plantations as refuges, where fire controls are of higher priority by humans. The role of corridors into external mammalian ranges is more crucial when considering the limitations of refuges for mammals to flee from the altered fire regimes in the BMC (Pardini et al., 2004).

Many studies throughout Australia have found that smaller fires across many locations allow for higher mammal survival rates than single extensive fires (Andersen et al., 2005; Bradstock, 2008; Kelly et al., 2012; Lawes et al., 2015; Pastro et al., 2011). The BMC's recent trend of many small fires seen in this study, provides some positive potential for the mammals' reactions to and survival in the recent fire regime changes. However, Griffiths et al. (2015) concluded that a high fire frequency affects mammal mortality more than the extent of fires. How fire affects each species, generally depends on various life traits (such as body size, dispersal capabilities, habitat choice and home-range) (Bendell, 1974; Griffiths & Brook, 2014a; Lawes et al., 2015). It seems that smaller study species with weak dispersal methods and smaller home-ranges were most directly threatened by 2017 (Griffiths & Brook, 2014a; Lawes et al., 2015). Higher fire frequencies, further from settlements may pose serious risks to medium-sized mammal populations. This is based on the assumption that the more isolated areas are deep within core zones, where generally more pristine mammalian habitats are sustained (Ishwaran et al., 2008; Pool-Stanvliet et al., 2017).

2.5.3 Conclusions and future recommendations

Conclusions

A strong contrast is revealed between land-cover change and fire regime shifts of mammalian habitats of the BMC. The overall shift to greater areas of vegetation from plantations and agriculture in the landscape is positive for mammalian habitats as it is a shift from two homogenous land-cover classes into a single heterogenous land-cover. However, fire regimes in the landscape conformed from diverse burn patterns to uniform burn patterns across the BMC, leading to homogenous fire disturbances (Andersen et al., 2005; Bradstock, 2008; Radford et al., 2015). The land-cover results alleviate concern over medium-sized mammalian habitat loss by human development, while the changes to the fire regime validate the original causes for concern over mammalian habitat damage. As the largest land-cover changes, vegetation gained was beneficial to medium-sized mammalian ecology in many ways. The loss of pine plantations has many of its own positive impacts on mammals and the ecosystem. The only concerns detected where human land-covers may threaten key mammalian habitats, were corridors into mammalian habitat ranges outside of the BMC. If the fire regime trends continue, as they have from 1956 to 2017, it is probable that many medium-sized mammals and their habitats' carrying capacities will be negatively affected (Bigalke & Willan, 1984).

On the selected scale of the study area, many of the primary drivers of reported fire regime and land-cover changes are proximate and influenced by the decisions of local government and stakeholders in the buffer zones. There was no indication that the changes in land-covers exacerbated fire regime impacts, and the shifts likely lessened the negative fire regime influences. CapeNature's enforcement of policies and land-owners' compliancy with policies ensured the gain of mammalian habitat. The majority of ignitions are sourced from human-activities and increases in the latter were the primary candidates for the increases in fire frequencies and area burned. Additionally, a decline in fire management resources and inconsistencies throughout the landscape affected what areas in the BMC burned more frequently.

Future action plans

The policies and attitudes of local government, CapeNature, WWF and many of the stakeholders of the BMC should be recognised for their positive improvements to medium-sized mammalian habitat. The two UNESCO Biosphere Reserves also need recognition for their roles in the structure and function of the landscape. Communication and management planning among stakeholders within and between the two biosphere reserves needs to be maintained. The enforcement of policies and land-owners compliance with policies, such as NEM:PAA (2003), must continue to prevent further encroachment into new or established habitats in the core or buffer zones of the BMC. Alternatively, and most ideally, to continue the shift from human land-covers to vegetation (preferably natural). More private properties along the buffer edges should be encouraged to enter into stewardship agreements (von Hase et al., 2010). Properties located at the BMC edges where potential habitat corridors are located would fulfil vital roles by entering into these stewardship agreements. Corridors within the study area, between nature reserves and leading to external mammalian habitat ranges need formal protection from land-cover change.

Implementation of integrated monitoring and management plans of human land-covers rehabilitation into fynbos and functional medium-sized mammalian habitats is advised, especially for felled pine plantations (of varying ages) (Houet et al., 2009; Ruiz, 2003). This must be applied to both privately-owned and government properties, with the aid of landscape management plans that are accessible and user friendly.

Wildfire management needs to be prioritised and all necessary resources (human resources, water reserves, fire-extinguishing tools, safety equipment, transport) sufficiently supplied by local government (Minas et al., 2012). Wildfire prevention at the sources of ignitions would be more cost effective and successful in achieving a stable and more sustainable fire regime, if implemented properly. This includes improving the safety standards of machinery, construction sites, modes of transport, power lines and other infrastructure exposed to buffer and core zones (CapeNature, 2017). All these structures need strict maintenance schedules and specific protocols in the event of a fire. Arson and accidental fires need to be treated more seriously as crimes, as they pose a risk to human lives, infrastructure damage and cost local municipalities huge amounts in resources to extinguish (Porter, 2009). In addition, this requires better education on the dangers and repercussions of fire for children, from early ages and for parents to enforce. Fire operation management on private properties needs continued monitoring and enforcement and adherence to policies and plans. Stakeholders on all buffer properties must be informed and educated in the fire policies of how to prevent wildfires and how to react to them. Resources for controlling fires should be made available on the properties. This would assist with faster responses, (where within the stakeholder’s control) while awaiting the arrival of trained fire fighters. Fire ignitions have strong correlations with human population sizes and are another reason for government to practise methods of reducing human population increase in the Western Cape (Archibald, 2016).

Future research

The South African Land-cover 1990 and 2013/2014 datasets original vegetation land-cover classes need to be more precisely defined to differentiate between stands of invasive species and natural vegetation (Tizora et al., 2016). Ideally, further studies would benefit from vegetation classes with multiple, more accurate-scoring sub-classes of the various vegetation-habitat types (such as lowland fynbos and renosterveld) (Von Hase et al., 2010). This would allow for better interpretation of the quality of medium-sized mammalian habitat in the BMC (and elsewhere) and identify what resources are available. Experimental studies are needed to assess each land-cover and fire's direct effects on medium-sized mammals in the Fynbos biome (Crooks, 2002; Parr & Chown, 2003). Critical mammalian habitat areas could then be mapped, for consideration in future development planning (CapeNature, 2017). Land-cover change assessments that consider these habitat types will be able to make more certain estimates as to how many faunal species would be affected in the BMC. To accurately conclude how this study's detected changes to the fire regime will affect medium-sized mammalian species, each of their fire ecologies in the Fynbos biome needs to be researched. This study reinforces the importance of analysing and managing the rehabilitation processes from pine plantation to natural habitats, as well as conversions from agriculture types, plantation and various built-up land-covers to vegetation. It is recommended that more in-depth landscape ecology assessments of fragmentation within the study area and the BMC as a fragment of the local leopard population's range in the Western Cape, be undertaken (Vogt et al., 2009). This would therefore consider the functionality of mentioned corridors.

2.6 Reference List

Andersen, A.N., Cook, G.D., Corbett, L.K., Douglas, M.M., Eager, R.W., Russell-Smith, J. et al. (2005). Fire frequency and biodiversity conservation in Australian tropical savannas: Implications from the Kapalga fire experiment. *Austral Ecology*, 30, 155-167.

Archibald, S. (2016). Managing the human component of fire regimes: Lessons from Africa. *Philosophical Transactions Royal Society B*, 371.

Archibald, S., Staver, A.C. & Levin, S.A. (2012). Evolution of human-driven fire regimes in Africa. *Proceedings of National Academy of Sciences*, 109(3), 847-852.

Archibald, S., Roy, D.P., van Wilgen, B.W. & Scholes, R.J. (2008). What limits fire? An examination of drivers of burnt area in Southern Africa. *Global Change Biology*, 15, 613-630.

Armstrong, A.J., van Hensbergen, H.J., Scott, D.F. & Milton, S.J. (1996). Are pine plantations "inhospitable seas" around remnant native habitat within South-western Cape forestry areas? *South African Forestry Journal*, 176(1), 1-9.

Auld, T.D. & Denham, A.J. (2001). The impact of seed predation by mammals on post-fire seed accumulation in the endangered shrub *Grevillea caleyi (Proteaceae*). *Biological Conservation*, 97(3), 377-385.

Barthelmess, E.L. (2014). Spatial distribution of road-kills and factors influencing road mortality for mammals in Northern New York State. *Biodiversity and Conservation*, 23(10), 2491-2514.

Beukes, P.C. (1987). Responses of grey rhebok and bontebok to controlled fires in coastal renosterveld. *South African Journal of Wildlife Research*, 17(3), 103-108.

Beier, P. (1995). Dispersal of juvenile cougars in fragmented habitat. *The Journal of Wildlife Management*, 59(2), 228-237.

Bendell, J.F. (1974). Effects of fire on Birds and Mammals. In T.T. Kozlowski & C.E. Ahlgren (Eds.), *Fire and Ecosystems* (pp. 73-150). New York: Academic Press Inc.

Benton, T.G., Vickery, J.A. & Wilson, J.D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, 18(4), 182-188

Bigalke, R.C. & Willan, K. (1984). Effects of fire regime on faunal composition and dynamics. *Ecological Effects of Fire in South African Ecosystems,* 48, 255-271.

Birtsas, P., Sokos, C. & Exadactylos, S. (2012). Carnivores in burned and adjacent unburned areas in a Mediterranean ecosystem. *Mammalia*, 76, 407-415,

Bond, W.J. & Keely, J.E. (2005). Fire as a global “herbivore”: The ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution*, 20, 387-394.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2001). A pragmatic approach to estimating the distributions and spatial requirements of the medium- to large-sized mammals in the Cape Floristic Region, South Africa. *Diversity and Distributions*, 7, 29-43.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2002). Estimated spatial requirements of the medium- to large-sized mammals, according to broad habitat units, in the Cape Floristic Region, South Africa. *Africa Journal of Range and Forage Science*, 19(1), 29-44.

Bowman, D.M.J.S., Balch, J., Artaxo, P., Bond, W.J., Cochrane, M.A., D’Antonio, C.M. et al. (2011). The human dimension of fire regimes on Earth. *Journal of Biogeography*, 38, 2223-2236.

Bradstock, R.A. (2008). Effects of large fires on biodiversity in south-eastern Australia: Disaster or template for diversity? *International Journal of Wildland Fire*, 17, 809-822.

Brodie, J.F., Giordano, A.J. & Ambu, L. (2014). Differential responses of large mammals to logging and edge effects. *Mammalian Biology*, 80, 7-13.

Cardillo, M., Purvis, A., Sechrest, W., Gittleman, J.L., Bielby, J. & Mace, G.M. (2004) Human population density and extinction risk in the world's carnivores. *PLoS Biology,* 2(7), 909-914.

Chan, C., van Vuuren, B.J. & Cherry, M.I. (2011). Fynbos fires may contribute to the maintenance of high genetic diversity in orange breasted sun birds (*Anthobaphes violacea*). *South African Journal of Wildlife Research*, 41(1), 87-94.

Chao, L. (2002). Estimation of fire frequency and fire cycle: A computational perspective. *Ecological Modelling*, 154, 103-120.

Chia, E.K., Bassett, M., Leonard, S.W.J., Holland, G.J., Ritchie, E.G., Clarke, M.F. et al. (2016). Effects of the fire regime on mammal occurrence after wildfire: Site effects vs landscape context in fire-prone forests. *Forest Ecology and Management*, 363, 130-139.

Clarke, M.F. (2008). Catering for the needs of fauna in fire management: Science or just wishful thinking? *Wildlife Research*, 35, 385-394.

Cochrane, M.A. & Laurance, W.F. (2002). Fire as a large-scale edge effect in Amazonian forests. *Journal of Tropical Ecology*, 18(3), 311-325.

Cochrane, M.A. & Laurance, W.F. (2004). Synergisms among fire, land use, and climate change in the Amazon. *Ambio*, 37, 522-527.

Converse, S.J., White, G.C., Farris, K.L. & Zack, S. (2006). Small mammals and forest fuel reduction: National-scale responses to fire and fire surrogates. *Ecological Applications*, 16(5), 1717-1729.

Crooks, K.R. (2002). Relative sensitivities of mammalian carnivores to habitat fragmentation. *Conservation Biology*, 16(2), 488-502.

DeFries, R., Hansen, A., Turner, B.L., Reid, R. & Liu, J. (2007). Land use change around protected areas: Management to balance human needs and ecological function. *Ecological Applications*, 17(4), 1031-1038.

Donaldson, R., van Niekerk, A., du Plessis, D. & Spocter, M. (2012). Non-metropolitan growth potential of Western Cape municipalities. *Urban Forumn*, 23, 367-389.

Duffell-Canham, A., Mortimer, G., Pence, G. & Pool-Stanvliet, R. (2017). Protected areas, biodiversity spatial planning and mainstreaming. In A.A*. Turner* (Ed.), *Western Cape Province state of biodiversity 2017* (pp. 19-38). Stellenbosch: Cape Nature Scientific Service.

Duncan, B.W., Schmalzer, P.A. (2004). Anthropogenic influences on potential fire spread in a pyrogenic ecosystem of Florida, USA. *Landscape Ecology,* 19, 153-165.

Elliot, L.P., Franklin, D.C. & Bowman, D.MJ.S. (2009). Frequency and season of fires varies with distance from settlement and grass composition in Eucalyptus miniata savannas of the Darwin region of northern Australia. *International Journal of Woodland Fire*, 18, 61-70.

Fahrig, L. (1997). Relative effects of habitat loss and fragmentation on population extinction. *The Journal of Wildlife Management*, 16(3), 603-610.

Fairbanks, D.H.K., Hughes, C.J. & Turpie, J.K. (2004). Potential impact of viticulture expansion on habitat types in the Cape Floristic Region, South Africa. *Biodiversity and Conservation*, 13, 1075-1100.

Fattebert, J., Dickerson, T., Balme, G., Slotow, R. & Hunter, L. (2013). Long-distance natal dispersal in leopard reveals potential for a three-country metapopulation. *South African Journal of Wildlife Research*, 43(1), 61-67.

Forsyth, G.G. & van Wilgen, B. (2008). The recent fire history of Table Mountain National Park and implication for fire management. *Koedoe*, 50, 3-9.

Fraser, M.W. (1990). Small mammals, birds and ants as seed predators in post-fire mountain fynbos. *South African Journal of Wildlife Research*, 20(2), 52-56.

Geary, W.L., Ritchies, E.G., Lawton, J.A., Healey, T.R. & Nimmo, D.G. (2018). Incorporating disturbance into trophic ecology: Fire history shapes mesopredator suppression by an apex predator. *Journal of Applied Ecology*, 55(4), 1594-1603.

GEOTERRAIMAGE 2014a. 1990 South African National Land-Cover Dataset. Data User Report and Metadata.

GEOTERRAIMAGE 2014b. 2013-2014 South African National Land-Cover Dataset. Data User Report and Metadata.

Golley, F. B., Ryszkowski, L., & Sokur, J. T. (1975). The role of small mammals in temperate forests, grasslands and cultivated fields. In F. B. Golley, K. Petrusewics, & L. Ryszkowski (Eds.). *Small mammals and their productivity and population dynamics* (pp. 223–242). Cambridge: Cambridge University Press.

Govender, T., Barnes, J.M. & Pieper, C.H. (2011). The impact of densification by means of informal shacks in the backyards of low-cost houses on the environment and service delivery in Cape Town, South Africa. *Environmental Health Insights*, 5, 23-52.

Griffiths, A.D. & Brook, B.W. (2014a). Effect of fire on small mammals: A systematic review. *International Journal of Wildland Fire*, 23, 1034-1043.

Griffiths, A.D. & Brook, B.W. (2014b). Fire impacts recruitment more than survival of small-mammals in a tropical savanna. *Ecosphere*, 6(6), 99.

Griffiths, A.D., Garnett, S.T. & Brook, B.W. (2015). Fire frequency matters more than fire size: Testing the pyrodiversity–biodiversity paradigm for at-risk small mammals in an Australian tropical savanna. *Biological Conservation*, 186, 337-346.

Gumbi, D.P. (2011). *The impact of change in climate, human demography, and other social factors on the fire regime of the Kogelberg Nature Reserve*. Unpublished doctoral dissertation, University of Kwazulu-Natal, South Africa.

Hall, L.S., Krausman, P.R. & Morrison, M.L. (1997). The habitat concept and a plea for standard terminology. *Wildlife Society Bulletin*, 25(10), 173-182.

Hannah, L., Roehrdanz, P.R., Ikegami, M., Shepard, A.V., Shaw, M.R., Tabor, G. et al. (2013). Climate change, wine and conservation. *Proceedings of National Academy of Sciences*, 110(17), 6907-6912.

Hansen, A.J. & DeFries, R (2007). Ecological mechanisms linking protected areas to surrounding lands. *Ecological Applications*, 17(4), 974-988.

Harris, L.D. & Scheck, J. (1991). From implications to applications: The dispersal corridor principle applied to the conservation of biological diversity. In D.A. Saunders & R.J. Hobbs (Eds.). *Nature conservation 2: The role of corridors* (pp.189-220). New South Wales: Surrey Beatty & Sons.

Henschel, P., Hunter, L.T.B., Coad, L., Abernethy, K.A. & Mhlenberg, M. (2011). Leopard prey choice in the Congo Basin rainforest suggests exploitative competition with human bushmeat hunters. *Journal of Zoology*, 285(1), 11–20.

Henzi, S.P., Brown, L.R., Barrett, L. & Marais, A.J. (2011). Troop size, habitat use, and diet of chacma baboons (*Papio hamadryas ursinus*) in commercial pine plantations: Implications for management. *International Journal of Primatology*, 32(4), 1020-1032.

Hitchcock, A., Cowell, C & Rebelo, T. (2012). The lost fynbos of Tokai Park. *Veld and Flora*, 98, 30-33.

Hoffman, M., Hilton-Taylor, C., Angulo, A., Böhm, M., Brooks, T. et al. (2010). The impact of conservation on the status of the world's vertebrates. *Science*, 330(6010), 1503-1509.

Holmes, P.M. & Richardson, D.M. (1999). Protocols for restoration based on recruitment dynamics, community structure, and ecosystem function: Perspectives from South African fynbos. *Restoration Ecology*, 7, 215-230.

Houet, T., Verburg, P. & Loveland, T. (2009). Monitoring and modelling landscape dynamics. *Landscape Ecology*, 25(2), 163-367.

Hradsky, B., Mildwaters, C., Ritchie, E., Christie, F., & Di Stefano, J. (2017). Invasive predator and native prey responses to a prescribed forest fire. *Journal of Mammalogy*, 98, 835– 847.

Iezzi, M.E., Cruz, P., Varela, D., De Angelo, C. & Di Bitetti, M.S. (2018). Tree monocultures in a biodiversity hotspot: Impact of pine plantations on mammal and bird assemblages in the Atlantic Forest. *Forest Ecology and Management*, 424, 216-227.

Ishwaran, N., Persic, A. & Tri, N.H. (2008). Concept and practice: The case of UNESCO biosphere reserves. *International Journal of Environment and Sustainable Development*, 7(2), 118-131.

Jacobson, A.P., Gerngross, P., Lemeris Jr., J.R., Schoonover, R.F. Anco, C., Breitenmoser-Würsten, C. et al. (2016). Leopard (*Panthera paradus*) status, distribution and research efforts across its range. *PeerJ*, 1974, 1-28.

Keeley, J.E. (2002). Native American impacts on fire regimes of the California coastal ranges. *Journal of Biogeography*, 29, 303-320.

Keeley, J.E., Pausas, J.G., Rundel, P.W., Bond, W.J & Bradstock, R.A. (2011). Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science*, 16(8), 406-411.

Kelly, L.T., Nimmo, D.G., Spence-Bailey, L.M., Taylor, R.S., Watson, S.J., Clarke, M.F. et al. (2012). Managing fire mosaics for small mammal conservation: A landscape perspective. *Journal of Applied Ecology,* 49, 412-421.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M, Boshoff, A.F. & Sims-Castley, R. (2003). Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112, 169-190.

Kintz, D.B., Young, K.R. & Crews-Meyer, K.A. (2006). Implications of land use/land cover change in the buffer zone of a national park in tropical Andes. *Environmental Management*, 38(2), 238-252.

Klar, N., Herrmann, M. & Kramer-Schadt, S. (2009). Effects and mitigation of road impacts on individual movement behaviour of wildcats. *Journal of Wildlife Management*, 73(5), 631-639.

Komarek, E.V. (1969). Fire and animal behaviour. Proceedings: 9th Tall Timbers Fire Ecology Conference. Tall Timbers Research Station, Tallahassee, Florida.

Kraaij, T. & van Wilgen, B.W. (2014). Drivers, ecology and management of fires in fynbos. In N. Allsopp, J.F. Colville & G. A. Verboom (Eds.), *Fynbos: Ecology, Evolution and Conservation of a Megadiverse Region* (pp. 47-72). Cape Town: Oxford University Press.

Kraaij, T., Baard, J.A., Cowling, R.M., van Wilgen, B.W. & Das, S. (2013). Historical fire regimes in a poorly understood, fire-prone ecosystem: Eastern coastal fynbos. *International Journal of Wildland Fire*, 22, 277-287.

Kraaij, T., Cowling, R.M. & van Wilgen, B.W. (2011). Paste approaches and future challenges to the management of fire and invasive alien plants in the new Garden Route National Park. *South African Journal of Science*, 107(9/10), 1-11.

Krebs, P., Pezzatti, G.B., Mazzoleni, F., Talbot, L.M. & Conedera, M. (2010). Fire regime: History and definition of a key concept in disturbance ecology*. Theory in Biosciences*, 129(1), 53-69.

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W. et al., (2001). The causes of land-use and land-cover change: Moving beyond the myths. *Global Environmental Change*, 11, 261-269.

Lawes, M.J., Murphy, B.P., Fisher, A., Woinarski, J.C.Z., Edwards, A.C. & Russel-Smith, J. (2015). Small mammals decline with increasing fire extent in northern Australia: Evidence from long-term monitoring in Kakadu National Park. *International Journal of Wildland Fire*, 24(5), 712-722.

Lawrence, G.E. (1966). Ecology of vertebrate animals in relation to Chaparral fire in the Sierra Nevada foothills. *Ecology*, 47(2), 278-291.

Legge, S., Murphy, S., Heathcote, J., Flaxman, E., Augustyn, J. & Crossman, M. (2008). The short-term effects of an extensive and high-intensity fire on vertebrates in the tropical savannas of the central Kimberley, Northern Australia. *Wildlife Research*, 35, 33-43.

Lindenmayer, D.B. & Hobbs, R.J. (2004). Fauna conservation in Australian plantation forests – a review. *Conservation Biology*, 119, 151-168.

Lindenmayer, D.B., Blanchard, W., MacGregor, D., Barton, P., Banks, S.C., Crane, M. et al. (2016). Temporal trends in mammal responses to fire reveals the complex effects of fire regime attributes. *Ecological Applications*, 26(2), 557-673.

Louw, W.J.A. (2010). General history of the South African Forest Industry: 2003-2006. *Southern African Forestry Journal*, 208(1), 79-88.

MacDonald, I.A.W., Graber, D.M., DeBenedetti, S., Groves, R.H. & Fuetes, E.R. (1988). Introduced species in nature reserves in Mediterranean-type climatic regions of the world. *Biological Conservation,* 44, 37-66.

Mack, M.C. & D'Antonio, C.M. (1998). Impacts of biological invasions on disturbance regimes. *Trends in Ecology and Evolution*, 13(5), 195-198.

Manandhar, R., Odeh, I.O.A. & Pontius Jr., R.G. (2009). Analysis of twenty years of categorical land transitions in lower Hunter of New Wales, Australia. *Agriculture, Ecosystems and Environment*, 135, 336-346.

Martins, Q. & Martins, N. (2006). Leopards of the Cape: Conservation and conservation concerns. *International Journal of Environmental Studies*, 63(5), 579-585.

McKillup, S. (2005). Statistics Explained: An Introductory Guide for Life Scientists. Cape Town: Cambridge University Press.

Merriam, G. (1991). Corridors and connectivity: Animal population in heterogenous environments. In D.A. Saunders, R.J. Hobbs (Eds.), *Nature Conservation: The Role of Corridors (pp. 133-142).* Chipping Norton: Surrey Beatty & Sons.

Minas, J.P., Hearne, J.W. & Handmer, J.W. (2012). A review of operations research (OR) methods applicable to wildfire management. *International Journal of Wildland Fire*, 21(3), 189-196.

Mostert, E., Gaertner, M., Holmes, P.M., Rebelo, A.G. & Richardson, D.M. (2017). Impacts of invasive alien trees on threatened lowland vegetation types in the Cape Floristic Region, South Africa. *South African Journal of Botany*, 108, 209-222.

Newmark, W.D. (2008). Isolation of African protected areas. *Frontiers in Ecology and the Environment*, 6(6), 321-328.

Nieman, W.A., Leslie, A.J., Wilkinson, A. & Wossler, T.C. (2019). Socioeconomic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa. *Journal of Nature Conservation*, 52, 1-7.

Novellie, P. (1987). Interrelationships between fire, grazing and grass cover at the Bontebok National Park. *Koedoe*, 30, 1-17.

Pardini, R., Nichols, E. & Püttker, T. (2018). Biodiversity response to habitat loss and fragmentation. In D.A. Dellasala & M.I. Goldstein (Eds.), *Encyclopedia of the Anthropocene* (pp. 229-239). Oxford: Elsevier.

Parr, C.L. & Chown, S.L. (2003). Burning issues for conservation: A critique of faunal fire research in Southern Africa. *Austral Ecology*, 28(4), 384-395.

Parrini, F. & Owen-Smith, N. (2009). The importance of post-fire regrowth for sable antelope in a Southern African savanna. *African Journal of Ecology*, 48, 526-534.

Pastro, L.A., Dickman, C.R. & Letnic, M. (2011). Burning for biodiversity or burning biodiversity? Prescribed burn vs. wildfire impacts on plants, lizards, and mammals. *Ecological Applications,* 21, 3238-3253.

Pool-Stanvliet, R. & Giliomee, J.H. (2013). A sustainable development model for the wine lands of the Western Cape: A case study of the Cape Winelands Biosphere Reserve. *AfriMAB*, 45-71.

Pool-Stanvliet, R. (2013). A history of UNESCO man and the biosphere programme in South Africa. *South African Journal of Science*, 109.

Pool-Stanvliet, R., Duffell-Canham, A., Pence, G. & Smart, R. (2017). The Western Cape spatial plan handbook. Stellenbosch: Cape Nature.

Porter, A. (2009). Arson: It's cost to insurers and the community. *Australian Journal of Forensic Sciences*, 19, 44-47.

Rabinowitz, A. (1990). Notes on the behavior and movements of leopard cats, *Felis bengalensis*, in a dry tropical forest mosaic in Thailand. *Biotropica*, 22(4), 397-403.

Racey, G.D. & Euler, D.L. (1982). Small mammal and habitat response to shoreline cottage development in central Ontario, Canada. *Canadian Journal of Zoology*, 60, 865-880.

Radford, I.J., Gibson, L.A., Corey, B., Carnes, K. & Fairman, R. (2015). Influence of fire mosaics, habitat characteristics and cattle disturbance on mammals in fire-prone savanna landscapes of the northern Kimberley. *PLoS ONE*, 10(6).

Ray, J.C., Hunter, L. & Zigouris, J. (2005). *Setting conservation and research priorities for larger African carnivores*. Working paper 24. New York: Wildlife Conservation Society.

Rebelo, A.J., Rebelo, A.G., Rebelo, A.D. & Bronner, G.N. (2019). Effects of alien pine plantations on small mammal community structure in a southern African biodiversity hotspot. *African Journal of Ecology*, 1-14.

Rentsch, D. & Damon, A. (2013). Prices, poaching and protein alternatives: An analysis of bushmeat consumption around Serengeti National Park, Tanzania. *Ecological Economics*, 91, 1-9.

Richardson, D.M. (1998). Forestry trees as invasive aliens. *Conservation Biology*, 12, 18-26.

Richardson, D.M., van Wilgen, B.W., Le Maitre, D.C., Higgens, K.B. & Forsyth, G.G. (1994). A computer-based system for fire management in the mountains of the Cape Province, South Africa. *International Journal of Wildland Fire*, 4(1), 17-32.

Richardson, D.M., Williams, P.A. & Hobbs, R.J. (1994). Pine invasions in the southern hemisphere: Determinants of spread and inaudibility. *Journal of Biogeography*, 21, 511-527.

Rouget, M., Richardson, D.M., Cowling, R.M., Lloyd, J.W. & Lombard, A.T. (2003). Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation*, 112, 63-85.

Rowe-Rowe, D.T. & Lowry, P.B. (1981). Influence of fire on small-mammal populations in the Natal Drakensberg. *South African Journal of Wildlife Research*, 12, 130-139.

Ruiz, R.F. (2003). *Alternative land uses to forestry in the Western Cape: A case study of La Motte Plantation*. Unpublished master's thesis, Stellenbosch University.

Sánchez-Cordero, V., Illoldi-Rangel, P., Linaje, M., Sarkar, S. & Peterson, A.T. (2005). Deforestation and extant distributions of Mexican endemic mammals. *Biological Conservation*, 126, 465-473.

Sauvajot, R.M. (1995). Conservation science in fire-prone natural areas. In J. E. Keeley & T. Scott (Eds.), *Brushfires in California: Ecology and Resource Management* (pp. 11–19). International Association of Wildland Fire, Fairfield.

Sauvajot, R.M., Buechner, M., Kamradt, D.K. & Schonewald, C.M. (1998). Patterns of human disturbance and response by small mammals and birds in chaparral near urban development. *Urban Ecosystems*, 2, 279-297.

Schwilk, D.W., Keeley, J.E. & Bond, W.J. (1997). The intermediate disturbance hypothesis does not explain fire and diversity pattern in fynbos. *Plant Ecology*, 132. 77-84.

Serieys, L.E.K., Bishop, J., Okes, N., Broadfield, J., Winterton, D.J., Poppenga, R.H. et al. (2019). Widespread anticoagulant poison exposure in predators in a rapidly growing South African city. *Science of the Total Environment*, 666, 581-590.

Smith, R.K., Jennings, N.V., Robinson, A. & Harris, S. (2004). Conservation of European hares *Lepus europaes* in Britain: Is increasing habitat heterogeneity in farmland the answer? *Journal of Applied Ecology*, 41, 1092-1102.

Sonestrom, D.A., Scanlon, B.R. & Zhang, L. (2009). Introduction to special section on impacts of land use change on water resources. *Water Resources Research*, 45.

Southey, D. (2009). *Wildfires in the Cape Floristic Region: Exploring vegetation and weather as drivers of fire frequency*. Unpublished master's thesis, University of Cape Town.

Spatial Planning (2018). Western Cape land-use planning: Rural guidelines. Draft report for public comment: January 2018, Cape Town.

Statistics South Africa (2018). *Mid-year population estimates 2018, Statistical Release P0302*, Statistics South Africa, Pretoria.

Stein, A., Athreya, V., Gerngross, P., Balme G., Henschel, P., Karanth U. et al. (2016). *Leopard Panthera pardus* [Online]. Retrieved August 24, 2018: https://www.iucnredlist.org/species/15954/102421779

Syphard, A.D., Radeloff, V.C., Hawbaker, T.J. & Stewart, S.I. (2009). Conservation threats due to human-caused increases in fire frequency in Mediterranean-climate ecosystems. *Conservation Biology*, 23(3), 758-769.

Tizora, P., Le Roux, A., Mans, G. & Cooper, A. (2016). *Land use and land cover change in the Western Cape Province: Quantification of changes & understanding of driving factors.* Unpublished paper delivered at the Seventh Planning Africa Conference on Making Sense of the Future: Disruption and Reinvention. Johannesburg, July 4-6.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I. & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters*, 8, 857-874.

University of Stellenbosch Business School (2018). Annexure A: 4IR Western Cape summary. The future of the Western Cape agricultural sector in the context of the 4th industrial revolution.

Van Hensbergen, H.J., Botha, S.A., Forsyth, G.G. & Le Maitre, D.C. (1992). *Do small mammals govern vegetation recovery after fire in fynbos? Fire in South African Mountain Fynbos*. Berlin: Springer-Verlag.

Van Wilgen, B.W. (2009). The evolution of fire and invasive alien plant management practices in fynbos. *South African Journal of Science*, 105, 335-342.

Van Wilgen, B.W., Forsyth, G.G., & Prins, P. (2012). The management of fire-adapted ecosystems in an urban setting: The case of Table Mountain National Park, South Africa. *Ecology and Society*, 17, 8.

Van Wilgen, B.W. (2013). Fire management in species-rich Cape fynbos shrublands. *Frontiers in Ecology and the Environment*, 11(1), 35-44.

Van, Wilgen, B.W. (2015). Plantation forestry and invasive pines in the cape floristic region: Towards conflict resolution. *South African Journal of Science*, 111, 1-2.

Van Wilgen, B.W., Forsyth, G.G., de Klerk, H., Das, S., Khuluse, S. & Schmitz, P. (2010). Fire-management in Mediterranean-climate shrublands: A case study from the Cape Fynbos, South Africa. *Journal of Applied Ecology*, 47, 631-638.

Vogt, P., Ferrari, J.R., Lookingbill, T.R., Gardener, R.H., Ritters, K.H. & Ostrapowicz, K. (2009). Mapping functional connectivity. *Ecological Indicators*, 9(1), 64-71.

Von Hase, A., Rouget, M. & Cowling, R.M. (2010). Evaluating private land conservation in the Cape lowlands, South Africa. *Conversation Biology*, 24(5), 1182-1189.

Wells, M.L., O'Leary, J.F., Franklin, J. Michaelsen, J. & McKinsey, D.E. (2004). Variations in a regional fire regime related to vegetation type in SanDiego County, California (USA). *Landscape Ecology* 19, 139–152.

Whelan, R.J. (1995). The ecology of fire. Cambridge University Press, Cambridge.

Willan, K. & Bigalke, R.C. (1981). The effects of fire regime on small mammals in S.W. Cape montane fynbos (Cape macchia). General Technical Report. PSW-58. Berkeley, California: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Wylie, D. (2016). *The urban edge: A spatial planning tool or device for land development management: A Western Cape perspective*. Unpublished master's thesis, University of Cape Town.

Zanne, A.E., Keith, B., Chapman, C.A. & Chapman, L.J. (2001). Protecting terrestrial mammal communities: Potential role of pine plantations. *East African Wild Life Society*, 39, 399-401.

Chapter 3

Medium-sized mammal occurrences and changes perceived by stakeholders on agricultural buffer properties

Brittany C. Schultz[1], Anita Wilkinson[2], Willem A. Nieman[1], Alison J. Leslie[1]

[1]Department of Conservation Ecology and Entomology, University of Stellenbosch, Matieland 7602, Western Cape, South Africa.

[2]The Cape Leopard Trust, P.O. Box 31139, Tokai, 7966, Cape Town, South Africa.

3.1 Abstract

The Protected Areas (PAs) of the Boland Mountain Complex (BMC), in the Western Cape harbour a relatively high diversity of medium-sized mammals. These mammals utilise agricultural buffer landscapes, which are the interface between human development and natural ecosystems. Anthropogenic activities and edge effects are prevalent along these agro-ecosystems and could act as population sinks and/or expose the PAs to a variety of negative impacts. This study aimed to determine whether perceived variations in distributions of and changes to medium-sized mammal abundances threaten the prey-base of local leopard *Panthera pardus pardus* populations. To do this, the objectives were to determine medium-sized mammal population distribution ranges and relative abundances and whether these mammals are displaying a population change across an agricultural buffer landscape in the BMC.

Due to a lack of historic population data, species presence-only and frequency of sighting data were gathered to estimate relative distribution-abundances. Structured interviews with agricultural labourers and management stakeholders were conducted, relying on their combined long-term Local Ecological Knowledge (LEK) of medium-sized mammal occurrence and abundance on private properties. Differences and changes in occurrence between different locations, farm characteristics and biosphere reserves were then investigated. Species occurrences varied between mountain zones (MZs) within the BMC, percentage farmed area, distances of properties from roads, settlements and PA boundaries. Twelve medium-sized mammal species were observed at significantly disproportionate abundances among various locations in the BMC. Distance from major roads, human settlements and PAs and the percentage land-use of properties influenced the presence of 14 mammal species. Caracal were the only species with a consistently reported stable population abundance in the BMC. Rock hyrax and baboon showed declines under specific conditions.

Suggestions are presented as to what species are of highest conservation concern. Grey rhebok *Pelea capreolus*, and hare *spp*. require priority research, monitoring and management. Feral dogs *Canis lupus familiaris* were detected as a prominent threat and knowledge of their sources, drivers and impacts need to be documented. Leopard should persist with the current high diversity and the high perceived abundances of certain mammal species available on agro-ecosystems. It is however, the direct mortality risks which leopard are exposed to when moving through these properties that needs strong consideration.

Keywords: Agro-ecosystems, Local Ecological Knowledge, Medium-sized mammals, relative-abundances.

3.2 Introduction

A high diversity of medium-sized mammals occurs throughout the core PAs of which many utilise the agricultural buffer landscapes of the BMC (Sinclair, 2003; Schoener, 1982). Each mammal species plays an important role in their ecosystem and is a potential prey item for leopard (Hayward et al., 2006; Hewitt & Miyanishi 1997). Predatory mammals' roles in ecosystems are to regulate other species and taxa's population sizes, which can indirectly affect vegetation and landscape structure (Sinclair, 2003). Other roles may contribute to seed dispersal, habitat engineering, disturbance (which drives vegetation factors), competitive, mutualistic or parasitic relationships with other species and many other effects (Hewitt & Miyanishi, 1997; Milton & Dean, 2000; Shiponeni & Milton, 2005). High mammal diversity ensures all these functions can occur and enhances the resilience of an ecosystem to both natural and anthropogenic disturbances (Quinn, 1986). Prey-base is a fundamental habitat resource to all carnivores and is the most common cause of concern over their survival, after direct mortality risks (Balme et al., 2010; Khan et al., 2018; Marker & Dickman, 2005). Prey availability is a key evolutionary and behavioural driver for carnivores (Owen-Smith & Mills, 2007). Multi-species studies in a habitat are more effective in understanding ecosystem functionality. These studies are thus important to determine natural population fluctuations and to identify when anthropogenic pressure drives a population change (Ĉervinka, et al., 2013; Hewitt & Miyanishi, 1997).

As in natural ecosystems, species utilise agro-ecosystems in a variety of ways - as corridors, refuges, population sources and sinks, to forage etc. (Ĉervinka, et al., 2013; Clark & Reeder, 2005). Thus, multi-scale and multi-species approaches are highly applicable to these anthropogenic biomes and their trophic levels (Ĉervinka, et al., 2013). Agricultural encroachment and anthropogenic threats contribute to natural pressures such as disturbance and resource competition, leading to mammals territories overlapping with farmed land (Fehlmann et al., 2017; Swanepoel et al., 2013). Agricultural land-uses have many impacts on surrounding natural ecosystems and are one of the leading causes of biodiversity loss at localized niche, landscape and climatic scales, worldwide (McLaughlin & Mineau, 1995; Rouget et al., 2003). Habitat loss (Lombard et al., 1997), fragmentation (Pardini et al., 2018), introduction of invasive species (Both 1989; Lewis et al., 2017), human-wildlife conflicts (Martins & Martins, 2006) and simplification of ecosystems are consequences of farms that negatively affect medium-sized mammal populations (Altrichter & Boaglio, 2004; Nelner & Hood, 2011). Farmlands do benefit some species that are generalists in their habitat selection (Park, 2014). Lower intensity agriculture with the presence of natural habitats can harbour a higher diversity of mammals (Matthiae & Stearns, 1981; Nelner & Hood, 2011; Tscharntke et al., 2005). Agro-ecosystems can act as buffer habitats to the adjacent PAs. These ecosystems are taking on more important ecological roles, as natural habitats are further isolated and encroached upon by agriculture (Clark & Reeder, 2005). Privately owned lands (such as the farms in these buffer habitats) are increasingly recognised for their vital roles in conservation, that need to be further researched (Fairbanks, 2002; Rouget et al., 2003).

There is a greater research focus on large mammals than on most medium-sized mammals in South Africa (Boshoff et al., 2002; Lloyd, 2000). In general, there is limited research on these smaller species, which usually occur at low densities and have shy habits, such as the Smith's red rock rabbit *Pronolagus rupestris*, honey badger *Mellivora capensis* or striped polecat *Ictonyx striatus* (Pettorelli et al., 2010). This knowledge deficiency is more prominent for populations in the Fynbos biome where population sizes are often regarded as low in comparison to other biomes (Kerley et al, 2003; Lloyd, 2000). There are a large number of studies focussing on the Fynbos biome's renowned floral diversity, but only a few focus on faunal diversity (Kerley et al, 2003).

An umbrella species is one that, if protected, its ecosystem and other species within it are too (Caro, 2003; Lambeck, 1997). Umbrella species are typically vulnerable to habitat disruption and have wide home-ranges that overlap other species ranges and habitat resources (Caro, 2003; Lambeck, 1997). Leopard therefore ideally fulfil this role in this study (Gehring & Swihart, 2002; Morrison et al., 2007; Sinclair, 2003). Leopard *Panthera pardus pardus* are opportunistic hunters utilising a diversity of species that must be maintained to support these predators (Hayward et al., 2006; Karanth & Chellam, 2009; Mann et al., 2019). The primary source of nutrients for leopard populations in the Western Cape are medium-sized mammals (Frőhlich, 2011; Martins et al., 2010; Norton et al., 1986; Ott et al., 2007; Rautenbach, 2010). Klipspringer *Oreotragus oreotragus*, rock hyrax

Procavia capensis, Cape grysbok *Raphicerus melanotis*, and Cape porcupine *Hystrix africaeaustralius* make up the majority of the diet of leopard in the Boland Mountain Complex (BMC) (Mann et al., 2019). Additionally, 14 other local species of medium-sized mammals have been recorded as prey items for leopard throughout the Western Cape, and it is likely that many others are consumed opportunistically (Braczkowski et al., 2012; Frőhlich, 2011; Mann et al., 2019; Martins et al., 2010; Norton et al., 1986; Ott et al., 2007; Rautenbach, 2010).

Mammal distribution data in the study area are outdated and only recently have some species that were previously thought locally extinct been declared present in the BMC. It is for this reason that initiatives such as the MammalMap virtual museum, which rely on citizen scientists, commenced (Animal Demography Unit, 2019). The number of mammal species that occur in the BMC is considerably high for an ecosystem consisting of rugged mountains, low-nutrient fynbos vegetation and that is anthropogenically isolated (Radloff et al., 2010; Rebelo et al., 2006). There are no systematic long-term records of these mammalian species abundances for the region, and those that have attempted to determine mammalian densities have found it too difficult by means of traditional count methods (Norton et al., 1986). Base-line prey population abundances, changes and diversity data are required for agro-ecosystems in the BMC to gain an understanding of the agricultural threats to mammals (Oberosler et al., 2017).

Making use of Local Ecological Knowledge (LEK) is an option to determine base-line mammal abundance for multiple species, with limited time and funds (Anadón et al., 2010). This method can account for the lack of historic research data (Leeney, 2015). It is ideal for low population densities over a large, diverse landscape where both mammalian and human ecological factors need to be considered (Anadón et al., 2010; White et al., 2005). It therefore enables conclusions and future priorities to be drawn at a more urgent pace (Anadón et al., 2010). This method was commonly and successfully used in studies on fishery stock abundances, however it has infrequently been practiced on terrestrial species (Azzurro et al., 2011; Pillay et al., 2011). Balme et al. (2014) stated that more engagement with local practitioners was needed in South Africa, particularly regarding leopard. This is also applicable for other mammal species in the BMC and personal surveys of multiple stakeholders encourages this. Folke (2004) highlighted how important LEK is in generating management decisions, in that stakeholders are exposed to and have direct impacts on local conservation. Stakeholders' decisions determine the conservation success in buffer zone ecosystems on a daily basis. Their knowledge can enable better ecosystem and sociological management practices (Folke, 2004). These gathered data were unattainable elsewhere, important, relevant and easy to keep consistent during the study (Anadón et al., 2010; Azzurro et al., 2011; Bencin et al., 2016).

The reliance on opportunistic human detection of mammals for LEK can create bias toward detecting those species that are more easily observed (Pearce & Boyce, 2006). Nocturnal mammal species, for example, should display lower perceived abundances than diurnal species in this study, because the interviewees were generally only present and active on the properties during daylight hours. For this reason, we cannot effectively compare how most different species were perceived across the landscape to one another. The below-mentioned methods were implemented in order to keep the research methods consistent to try alleviating bias. By maintaining consistent methods of interviewing across all farms and interviewees the results for each species will be comparable to the same species throughout the BMC. It can be assumed that species that are more elusive to humans, such as nocturnal species or those that are perceived as less charismatic, will probably have higher relative-abundances than this study's results will show (Boakes et al., 2010; Nyhus et al., 2003).

Questionnaires with a closed-ended format guide response to a specific range of answers available are recommended (Schuman & Scott, 1987). By using close-ended questionnaires the study allows for responses that are mutually more understandable for interviewer and interviewee (White et al., 2005), and provides for more easily comparable results that can be interpreted as quantitative rather than qualitative (Driscoll et al., 2007). When limited data is available presence-only data can determine species vulnerability, historical distribution and conservation status (Pearce & Boyce, 2006). Presence-only data from stakeholders forms a dimension of analysing perceived species abundance and presents some important knowledge of an ecological system (Pearce & Boyce, 2006). The time-limit with regards to each participants memory is difficult to predict and can often affect the accuracies of data (Pearce & Boyce, 2006). For this mentioned-reason, this study did not have a set time-scale when questioning changes, but instead opened it up as qualitative data. As Schuman & Scott (1987) proposed, qualitative accounts give better overall descriptions of threats or concerns. Open-ended questions also reveal other aspects of the landscape that could not have predicted (Anadón et al., 2009).

The study area was selected in relation to the BMC which is a United Nations Environmental, Educational, Scientific and Cultural Organization (UNESCO) World Heritage Site and Protected Area (PA) in which the leopard is the apex predator. It is made up of seven PAs, surrounded by privately-owned agricultural properties and patches of human development/settlements. These settlements are some of the fastest and largest developing towns in the Western Cape (Spatial Planning, 2018; Statistics South Africa, 2018; Tizora et al., 2016) (Figure. 3.1).

The aim of this study was to determine medium-sized mammal population distributions, relative abundances and whether they are displaying a change over time or variation across an agricultural buffer landscape in the BMC. This is a means to determine how great of a threat prey-loss is to leopard in the BMC's agro-ecosystems. The primary objectives were:

i) To document perceived medium-sized mammal population presences and frequencies of sightings in order to estimate the relative abundance-distributions.

ii) To determine what aspects of agro-ecosystems in the BMC are driving variations in abundances of mammals (locations in the landscape, distances from features, land-use allocations).

iii) To highlight what mammalian populations and habitat ranges are perceived to be threatened and what threats are present.

iv) To provide data on where, how and what mammal populations are priorities to allocate future research, monitoring, management and legislative resources to.

3.3 Methodology

3.3.1 Study area

The study area is focused around the BMC and ranges within it that are accessible to local leopard populations. This area is overlaid by two adjoining UNESCO biosphere reserves, namely: the Kogelberg Biosphere Reserve (KBR) and the Cape Winelands Biosphere Reserve (CWBR) in the Western Cape, South Africa. The BMC displays Mediterranean climate patterns (Goldblatt & Manning, 2002). The area falls within the Fynbos Biome and naturally contains steep mountain slopes covered in fynbos and lower plains of mountains covered in lowland fynbos and renosterveld (Boshoff et al., 2001; Goldblatt & Manning, 2002; van Wilgen, 1987). Large expanses of the lowland fynbos and renosterveld have been replaced with agriculture and other human land-uses (Boshoff et al. 2001; Rouget et al., 2003).The study focused on privately owned farms within buffer agricultural zones. Surveyed properties ranged in size from five ha to 4107ha. These farms were operational subsistence and commercial properties. The percentage of land-use that was transformed from natural on these properties varied greatly between 0 and 100%. These transformations were mostly agricultural, including: vineyards, apple, pear, peach, pomegranate, plum, nectarine, olive, granadilla, kiwi and fynbos orchards, sweet potato, wheat, nut, cover crop, trout, extensive and intensive game, sheep, goat, cow, horse and pig farming. Forestry, roads, dams, event fields and other infrastructure were also included as transformed land. Many farms had staff villages and tourism features, for example, restaurants and accommodation facilities.

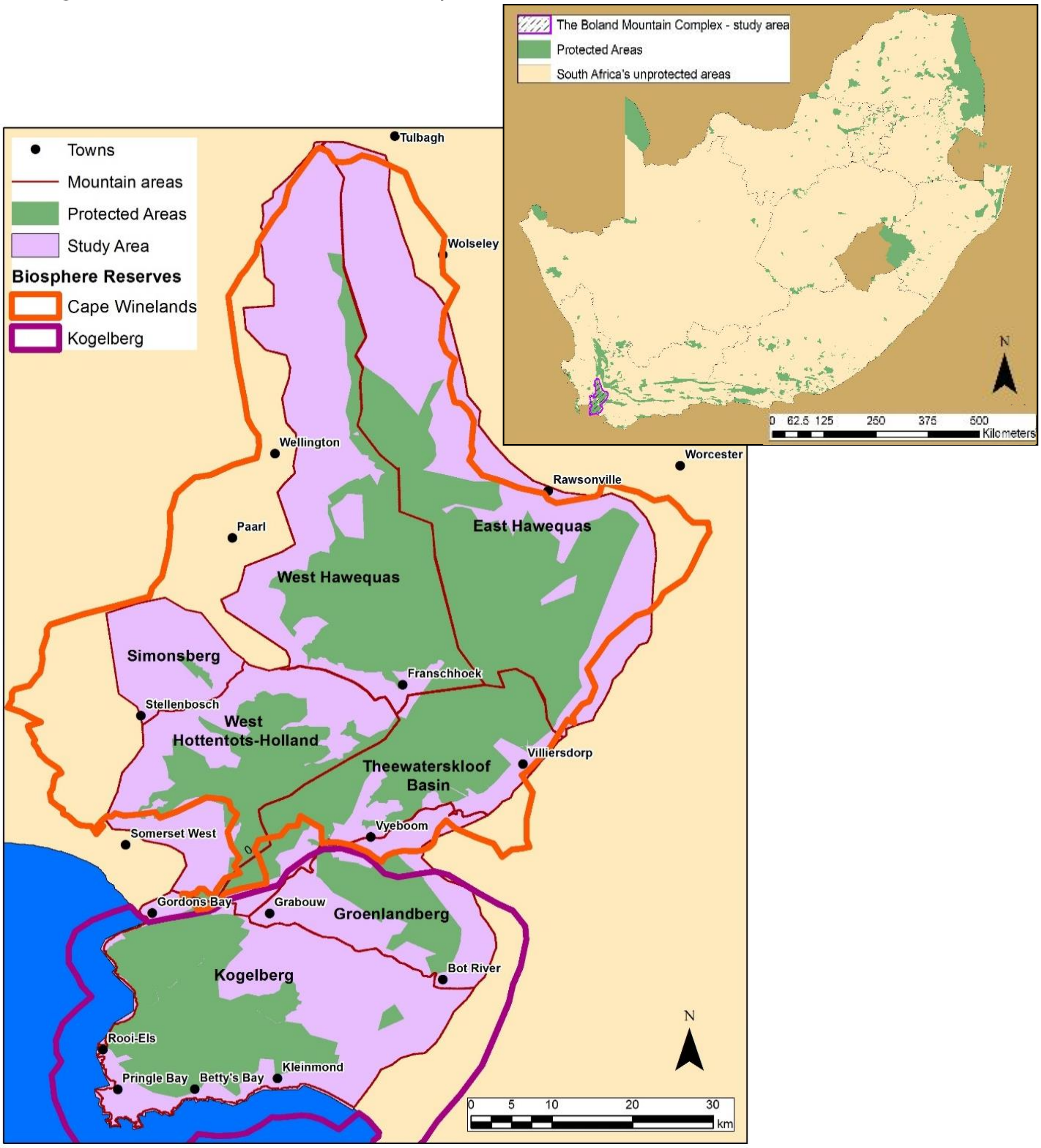


Figure 2.1: Location of the focal study area within the Boland Mountain Complex location in the Western Cape Province, South Africa and its two biosphere reserves and seven mountain zone divisions amongst protected areas and towns.

3.3.2 Data Collection

LEK is a data collection method that is open to human bias (Anadón et al., 2009; Boakes et al., 2010; White et al., 2005). Common errors during the interview process that are relevant to the data include poorly defined locations, sizes of areas, relevant timeframes and study species identification (Anadón et al., 2009; Bernard et al., 1984; Jones et al., 2008). The way in which the interview is carried out can drive some strong miscommunications especially when changing qualitative data into quantitative, which interviews in person that include open questions alleviate (Anadón et al., 2009; Jones et al., 2008). The methods described below were therefore followed with strict consistency between all properties and interviewees.

Data collection only took place on private properties that fell within agricultural buffer zones on the periphery of PAs. There were two questionnaires, one for each of the relevant participant groups on the properties: 1) farmers/managers and 2) farm labourers. Interview routines were set and kept consistent for each farm and interviewee. Within each of the two stakeholder groups the interviewees respectively fulfil similar roles in the landscape, i.e. managers or owners generally have similar actions and job descriptions on each farm as one another and labourers have similar hours and job descriptions as one another. Data collection was therefore consistent for comparison within each sample group and between the considered variables (location, biosphere reserve, distance from feature, land-use percentage).

Ethical clearance was granted by the Human Research (Humanities) Council, Stellenbosch University, (Reference: SU-HSD-004696). Prerequisite requirements for properties to be included in the study were that they should be located adjacent to core PAs, be larger than five ha in size and should employ permanent labourers who are exposed to the agro-ecosystem habitats on the properties. Of those farms that meet the criteria in the BMC, 42% of farms were selected through random stratified sampling. Appointments were scheduled with farm owners or managers telephonically. Criteria for participating labourers were that they had been present on the property for a long period of time and primarily work outdoors. The average number of years participants were present and worked on the properties was 16.9 years. Between July 2017 and May 2018, 99 farms were visited, where a total of 99 farm managers or owners, and 299 labourers were interviewed. Interviewees were briefed on the study purposes, shown and read a consent form which was confirmed verbally and signed by the interviewee in order for the interview to commence (Jones et al., 2008) (Appendix 3.3). Structured interviews were then conducted in person and privately with a manager or owner and two to five labourers on each property. The interviewer was kept consistent for all 398 interviews.

The structured questionnaire with 15 questions was developed, following the question formatting from Bencin et al. (2016) and incorporating suggestions from White et al. (2015). For both participant types, these questions pertain to frequency of mammal sightings (Appendix 3.1 & Appendix 3.2). Farm managers/owners were additionally presented with questions about the farm history/background and staff management (Appendix 3.2). Most questions were closed-ended, except those that asked for people's specific opinions as to "why" they gave a particular answer (open-ended question). When answers to semi-open ended questions were unclear, further prompting was applied. For example, when asked how often the participant sighted an animal per year and they answered, "a few times" they were prompted to give a specific number, "twice, three times, four times, five times, more?" Interviews were conducted in English or Afrikaans depending on the interviewee's preference. Questions and routines of interviews were kept consistent and written on printed questionnaire pages, with an additional space for extra qualitative data. A set of identification cards with images of each species were presented to the interviewee in order to determine which species they had or had not seen on the property (Burton et al., 2011) (Appendix 3.4). This visual aid helped prevent identification uncertainty due to colloquial names for species. For each species an interviewee had observed, 11 standard questions followed. Interviews lasted between 10 and 45 minutes each and two to three farms were visited in a day. Most of the answers were converted into quantitative data by means of binary format, scales or relative ratios. Species that were difficult to differentiate between, were grouped together. These included the large-spotted, *Genetta tigrina,* and small-spotted genets, *Genetta genetta,* as genets *Genetta* and the scrub hare *Lepus saxatilis* and Cape hares *Lepus capensis,* as hares *Lepus*.

3.3.3 Data Analysis

Data analyses were run for presence-only and frequencies of sightings and perceived population changes of each mammalian species. The presence-only data were converted into binary data of "1" for present or "0" for not sighted. Frequency of sightings were converted into the number of days in a year that a species was generally seen. This involved converting answers such as "daily" to "365 days per year," "weekly" to "52 days per year", "twice weekly" to "104 days per year," "monthly" to "12 days per year" etc. which was a process kept consistent for all answers and farms. Changes in species population data were used as categories of "increase", "decrease" or "stable". Those that answered "uncertain", showed uncertainty or met characteristics that affected their ability to accurately answer were excluded. Species profiles of the percentage of farms they were observed on were drawn up as well as the average frequencies at which they were observed per year across farms. The data were visualised on the entire study area, the two biosphere reserves and the seven MZs. MZs were decided according to where the sampled farms locations clustered near one another. These MZs were then outlined according to mountains that potentially reflected natural movement barriers to mammals and roads or settlements as barriers that reflect human management divisions. The seven MZs were West Hottentots-Holland, Simonsberg, West Hawequas, East Hawequas, Theewaterskloof Basin, Groenlandberg and Kogelberg MZs (Figure 3.1). Species “Relative Abundance-Distribution Profiles” were drawn up (using Microsoft Excel, 2017) on the study area level, biosphere reserve level and MZ level. The “Relative Abundance-Distribution Profiles” were estimated using the accumulation of the percentage of farms which had observed a species as present on the property multiplied by the average percentage of days per year which the species was sighted across the farms.

Presence-only data

Correspondence analyses were performed to test whether the percentage of farms where species reported occurrence differed significantly between BRs and between MZs, using Statistica version 13.3. (TIBC Software Inc, 2017). For those species that displayed convergent patterns, further Chi-squared analyses and Fisher’s exact test (to account for low sample sizes) were performed. Similarly, a Chi-squared analysis and Fisher’s exact were run when assessing the categorical changes to populations against these same area variables. Species observed on less than 5% of farms were excluded from the above analysis (i.e. Aardvark *Orycteropus afer*, aardwolf *Proteles cristata*, bushbuck *Tragelaphus sylvaticus*, fallow deer *Dama dama* and bush pig *Potamochoerus larvatus*).

All distance variables (distance from settlement, major roads and boundaries of PAs) were measured using ArcGIS’s ArcMap 10.4.1. (ESRI, 2015). Settlements were defined as human settlements with high densities of built-up structures. Major roads were those general public, tarred roads (off the private farm properties). Boundaries of the PAs were those that were officially defined on CapeNature’s data systems and may not show any defined physical barrier (like a fence) on the ground. Measurements were taken from the main offices/sheds on each farm, because these were the assumed centralised point that all respondents would access on the farm, regardless of each individual’s specific duties and movements on the properties. Distances were measured from the nearest edges of all major roads, PA and settlement boundaries, using the geodesic method of the “Near” tool. The boundary of the PA was sourced from CapeNature (2014) and roads were clipped from the shape file of major roads of South Africa and Lesotho (Open Database 1.0, 2014). The distances to human settlements were measured to those detected on the Department of Environmental Affairs 2013/2014 land-cover raster dataset, including all farm housing and infrastructure that clustered together enough that they represented the majority of a 30 by 30m cell (GEOTERRAIMAGE, 2014b).

Frequency of sightings

One-way Analysis of Variances (ANOVA's) were run for each species for the continuous variable "sightings per year" against the MZs and biosphere reserves. Significance was determined using Least Square means p-value. Where the data did not meet the assumptions of homogeneity, the Welch Test was run as it gives more accurate significance ratings (Keselman et al., 2004). ANOVA's with the Least Square means were used to examine species presence against the distance variables (from settlements, roads and protected boundaries) and against the percentage natural vegetation land-cover (McKillery, 2005). Levene's tests were run on the distance variables prior to these ANOVA's to confirm whether parametric tests may or may not be used (Gastwirth et al., 2009). For those Levene's tests which resulted in a $p<0.05$, a non-parametric Mann-Whitney U test was used to assess significance in these ordinal data. For the same results from the Levene's test, Games-Howell post hoc tests were used to determine where significant differences between the various distances of farms from the above-mentioned factors existed (Shingala & Rajyaguru, 2015). Some species were excluded from the frequency of sightings analysis due to low incidence ($n<10$) and uncertainties of sightings (made clear by the interviewees stating so), including: African wildcat *Felis silvestris lybicat*, bat-eared fox *Otocyon megalotis*, aardvark, aardwolf, feral cat, bushbuck, fallow deer, Smith's red rock rabbit, polecat, bush pig and feral pig *Sus scrofa*.

Both Spearman's Rank Order Correlation and Pearson correlations were run to test for a significant correlation between each species' yearly sighting frequency and four abiotic variables, i.e.: distance from PA boundaries, distance from settlements, distance to nearest major roads and the percentage natural land-use on farms. Both tests were performed due to the large sample size ensuring normality and Shapiro-Wilk tests were run to confirm this. Inverse Distance Weighted (IDW) Interpolations (ESRI, 2015) were created for each species sightings per days in the year, using ArcGIS's ArcMap 10.4.1. (ESRI, 2015). The interpolations estimated the relative abundances of each species/the perceived frequencies that they were sighted per year at points and in areas where no data were sampled. No barriers were included in the interpolation analysis. IDW interpolations are another visual representation of the relative abundances of each mammal species in the landscape. Using a 12km radius to incorporate those farms that clustered together, the spatial autocorrelation was run on the assumption that values closer to one another were better related. Closer points of sightings potentially refer to the same individuals or populations of a species.

3.4 Results

3.4.1 Presence-only and frequency of sightings

Mammals that were most widespread and present on the highest percentage of farms in the

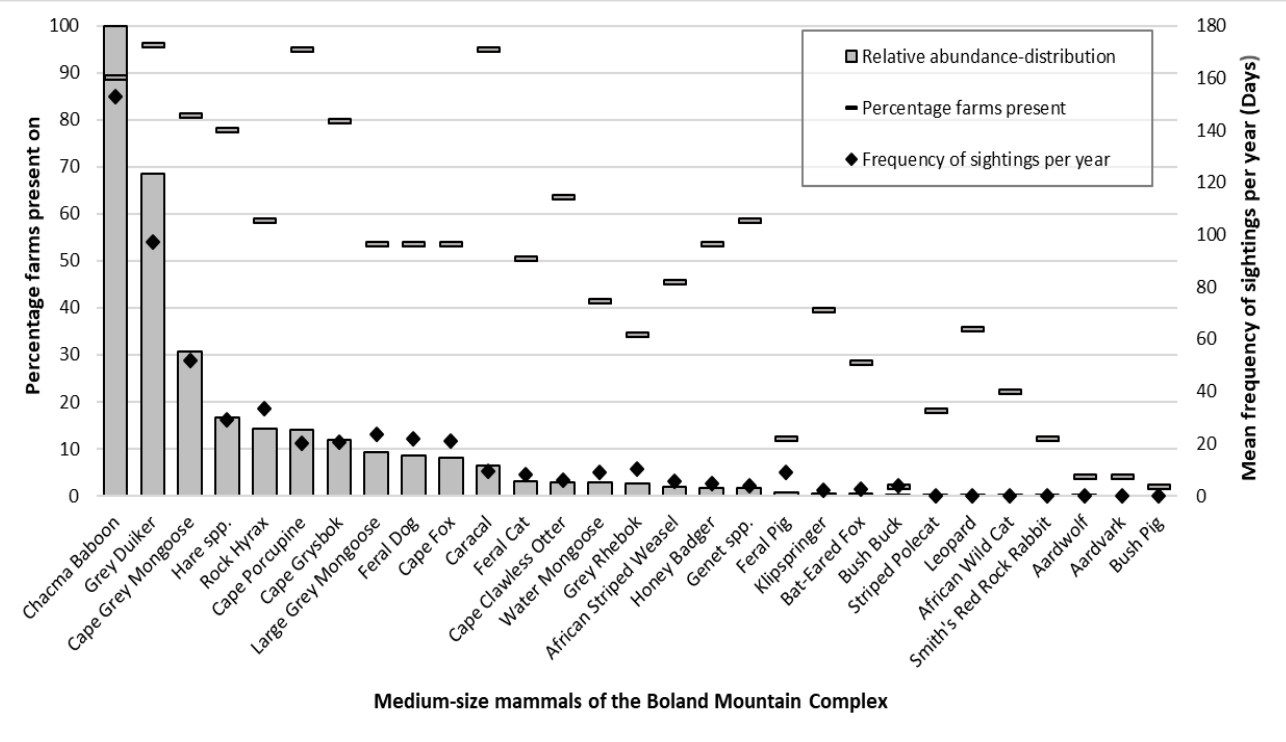


Figure 3.2: Estimated Relative Abundance-Distribution Profile of medium-sized mammals in the Boland Mountain Complex, based on the percentage farms that each was present on (left axis) and the mean frequency (number of days in a year) that each species was sighted (right axis).

BMC were common duiker *Sylvicapra grimmia* (95.96%), caracal *Caracal caracal* (94.95%), Cape porcupine *Hystrix africaeaustralis* (94.95%), chacma baboon *Papio ursinus* (88.89%), Cape grey mongoose *Galerella pulverulenta* (80.81%), Cape grysbok *Raphicerus melanotis* (79.8%) and hare (77.78%) (Figure. 3.2). Species that were sighted on the lowest percentage of farms in the BMC were the bush pig, fallow deer and bushbuck (2.02%), followed by the aardvark and aardwolf which were seen on equally few farms (4.04%). Other species seen on less than 50% of farms were Smith's red rock rabbit *Pronolagus rupestris* (12.12%), feral pig (12.12%), striped polecat (18.18%), African wildcat *Felis silvestres lybica* (22.22%), bat-eared fox (28.28%), grey rhebok *Pelea capreolus* (34.34%), leopard (35.35%), klipspringer (39.39%), water mongoose *Atilax paludinosus* (41.41%) and African striped weasel *Poecilogale albinucha* (45.45%). Other widespread species seen on over 50% of farms were feral domestic cat *Felis catus* (50.51%), large grey mongoose *Herpestes ichneumon* (53.54%), honey badger *Mellivora capensis* (53.54%), feral dog *Canis lupus familiaris*

(53.54%), Cape fox *Vulpes chama* (53.54%), genet (58.59%), rock hyrax (58.59%) and the Cape clawless otter *Aonyx capensis* (63.64%) (Figure. 3.2).

Species occurred on similar percentages of farms, in similar rank-orders in each of the biosphere reserves. Duiker, caracal, porcupine, baboon, Cape grey mongoose, grysbok and hares were sighted on the most farms in the KBR (Table. 3.1). Rankings of percentage farms presence in the CWBR were highest for duiker, caracal, porcupine, baboon, Cape grey mongoose, grysbok and hare (Table. 3.1).

Feral dogs (in packs and as individuals) were the third most frequently seen species per year in the CWBR and ranked 11th most frequently seen in the KBR. Grysbok were the sixth most frequently seen in the CWBR and the 15th in the KBR. Both weasels and badgers were seen more frequently in the CWBR (5 per year) than KBR. Hares were the third most frequently seen species per year in the KBR and ranked seventh in the CWBR. Cape Foxes were seen frequently in the KBR but rarely in the CWBR. Rhebok too were seen more regularly in the KBR than CWBR, ranking sixth and twentieth respectively (Table. 3.1).

Table 3.1: Rankings of each species' average frequency of sighting and percentage farms that they occurred on for the entire study area, the Kogelberg Biosphere Reserve (KBR) and Cape Winelands Biosphere Reserve (CWBR).

Species	**Rank of percentage of farms each species occurred on**			**Rank of average frequency that each species was sighted per year**		
	Study Area	**KBR**	**CBR**	**Study Area**	**KBR**	**CBR**
Duiker	1	1	1	2	2	2
Caracal	2	2	1	12	17	15
Porcupine	2	2	2	10	9	8
Baboon	3	2	3	1	1	1
Cape grey mongoose	4	2	4	3	4	5
Grysbok	5	3	5	9	15	6
Hare	6	4	6	5	3	7
Otter	7	5	7	16	12	11
Rock hyrax	8	5	7	4	7	4
Genet	8	5	8	19	13	19
Cape fox	9	7	9	8	5	10
Feral dog	9	7	10	7	11	3

Honey badger	9	6	11	18	20	14
Large grey mongoose	9	7	11	6	8	9
Feral cat	10	8	12	15	10	16
Striped weasel	11	8	13	17	18	11
Water mongoose	12	9	14	13	16	13
Klipspringer	13	9	15	22	14	21
Leopard	14	10	15	26	22	24
Rhebok	15	11	16	11	6	20
Bat-eared fox	16	11	17	21	19	17
Wildcat	17	11	18	25	23	23
Polecat	18	12	19	23	21	25
Feral pig	19	14	20	14	23	18
Rabbit	19	13	21	24	23	22
Aardwolf	20	15	22	27	23	25
Aardvark	20	15	22	28	23	25
Bushbuck	21	15	23	20	23	25
Fallow deer	21	15	23	29	23	25
Bush pig	21	15	24	29	23	25

3.4.2 Variations in perceived abundance between locations

Baboons were reported at significantly lower frequencies in Simonsberg MZ than anywhere in the BMC (Weighted Means Welch test F(6.0,30.9)=19.21, p<0.01) (Figure. 3.3A).

Hares had significantly less frequent mean sightings per year in the CWBR (19 per year) than the KBR (50 per year) (LS means: F(1.95)=5.2901, p=0.02) (Figure. 3.4B). They were also reported on a significantly lower percentage of farms in the West Hottentots-Holland MZ (52%) (Chi-squared (df=6)=20.47, p=0.002, Fisher exact p=0.01) and at lower frequencies (Welch test F(6.0,30.4)=2.66, p=0.03).

Rock hyraxes were reported on a significantly lower percentage of farms in the Groenlandberg MZ (18%) than the remaining six MZs in the BMC (Chi-squared (df=6)=15.79, p=0.015, Fisher exact p=0.02) (Figure. 3.5A).

Grysbok were reported on a significantly lower percentage of farms in the Groenlandberg MZ (55%) than the rest of the study area's MZs (Chi-squared (df=6)=13.13, p=0.041) (Figure. 3.6A).

Large grey mongoose were reported present on a significantly lower percentages of farms in Simonsberg (30%) and West Hottentots-Holland MZs (35%) than the remaining five MZs (Chi-square(df=6)=15.73, p=0.015) (Figure. 3.6B).

Feral Cats were reported present on significantly higher percentages of farms in Theewaterskloof Basin (89%), Groenlandberg (91%) and East Hawequas MZs (70%) than the remaining four MZs (Chi-squared (df=6)=23.13, p<0.001) (Figure. 3.8B).

Water mongoose were sighted at significantly higher frequencies in the East Hawequas MZ (72 per year) than all other six MZs (LS mean F(6,92)=3.2186, p=<0.01) (Figure. 3.9B)

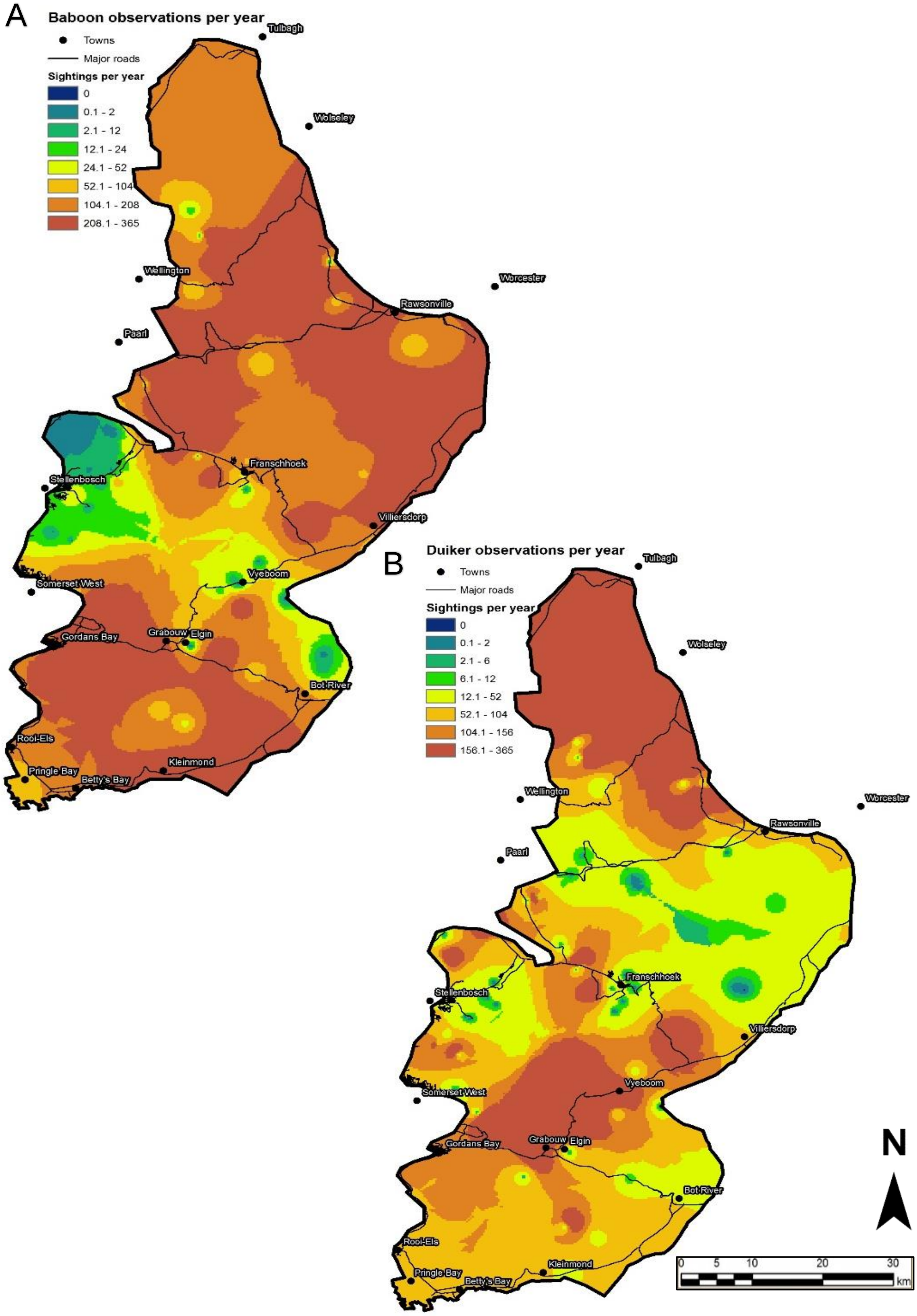


Figure 3.3: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Chacma baboon, B: Common duiker).

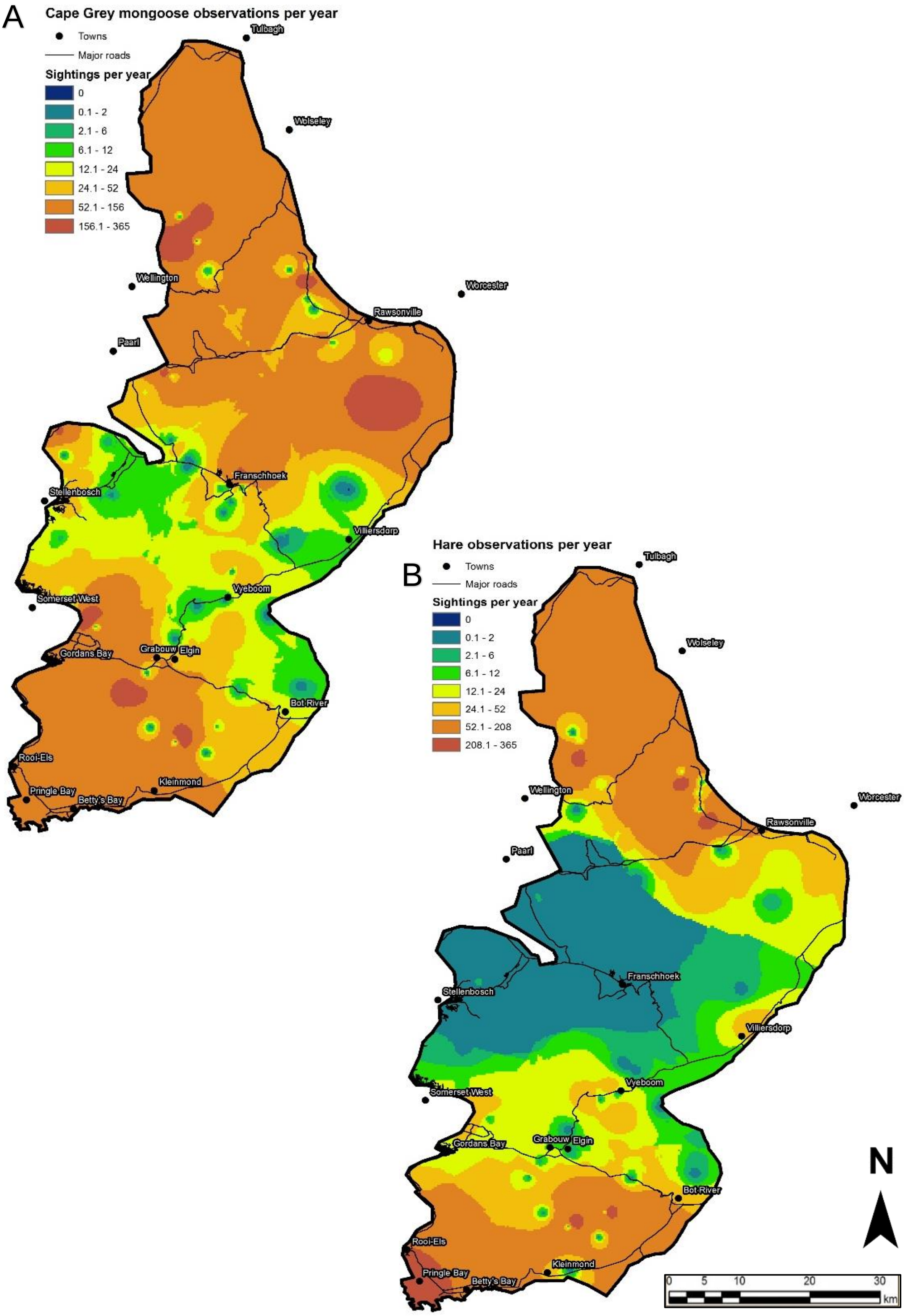


Figure 3.4: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Cape grey mongoose, B: Hare spp.)

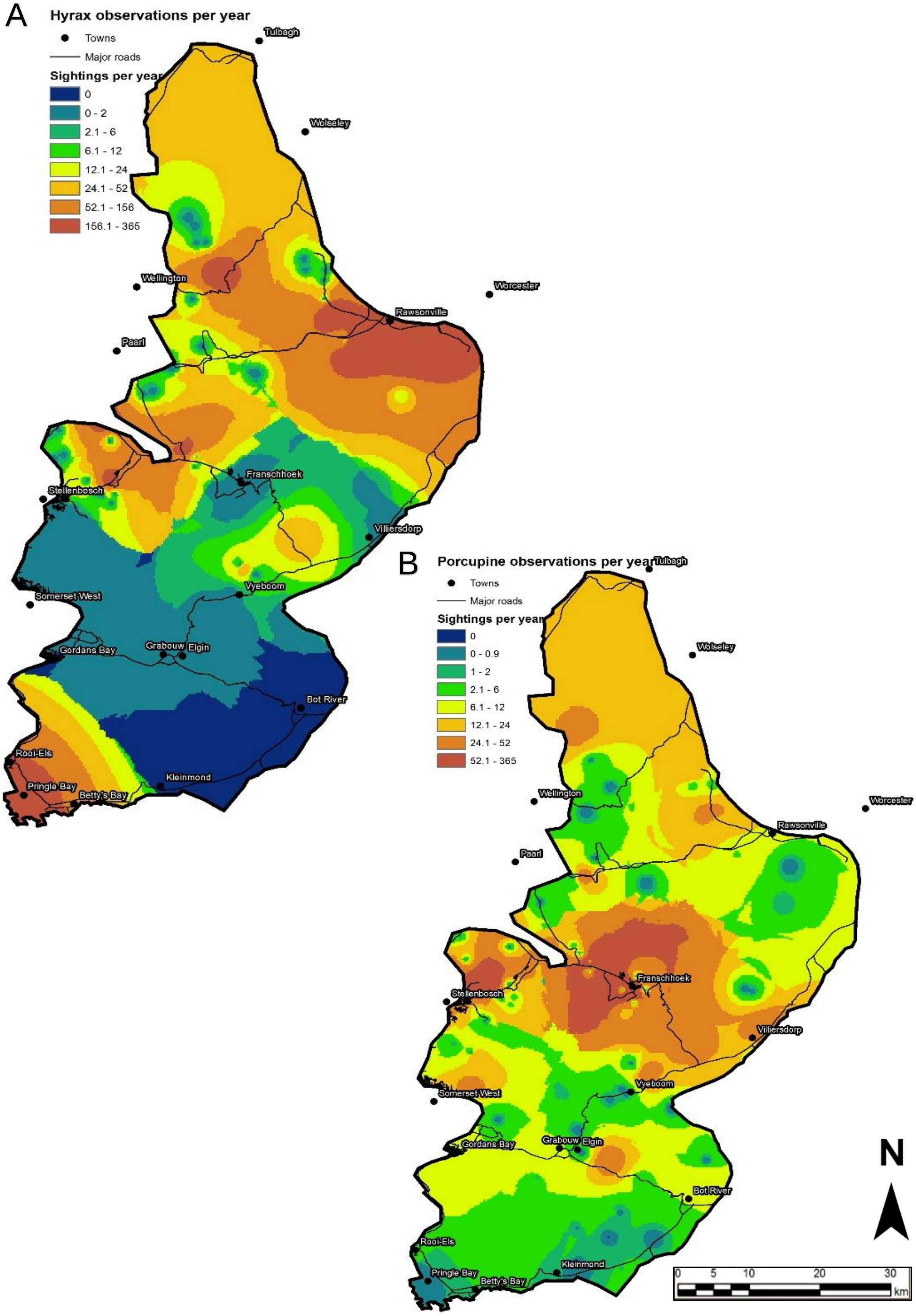


Figure 3.5: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Rock hyrax B: Porcupine).

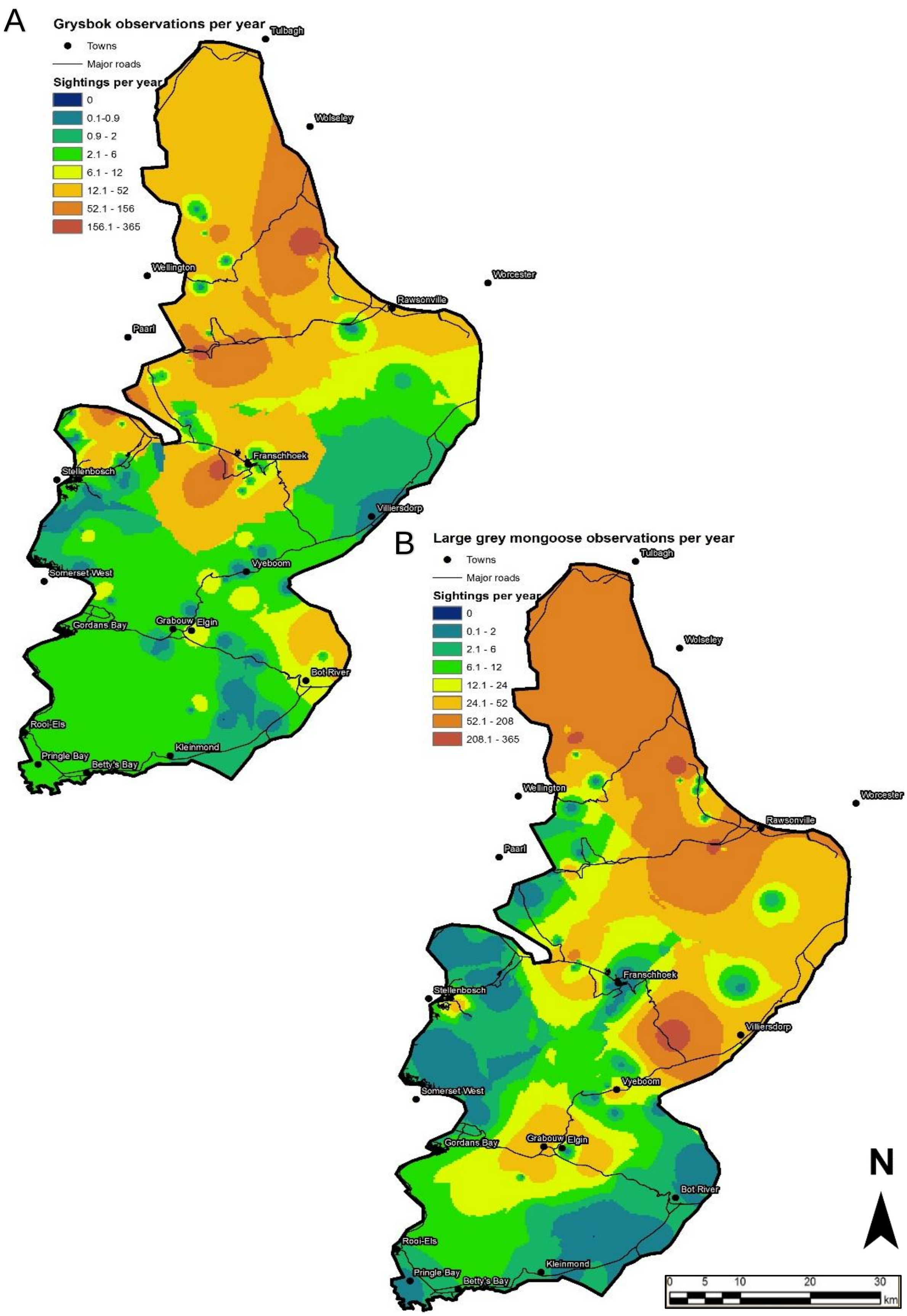


Figure 3.6: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Cape grysbok B: Large grey mongoose).

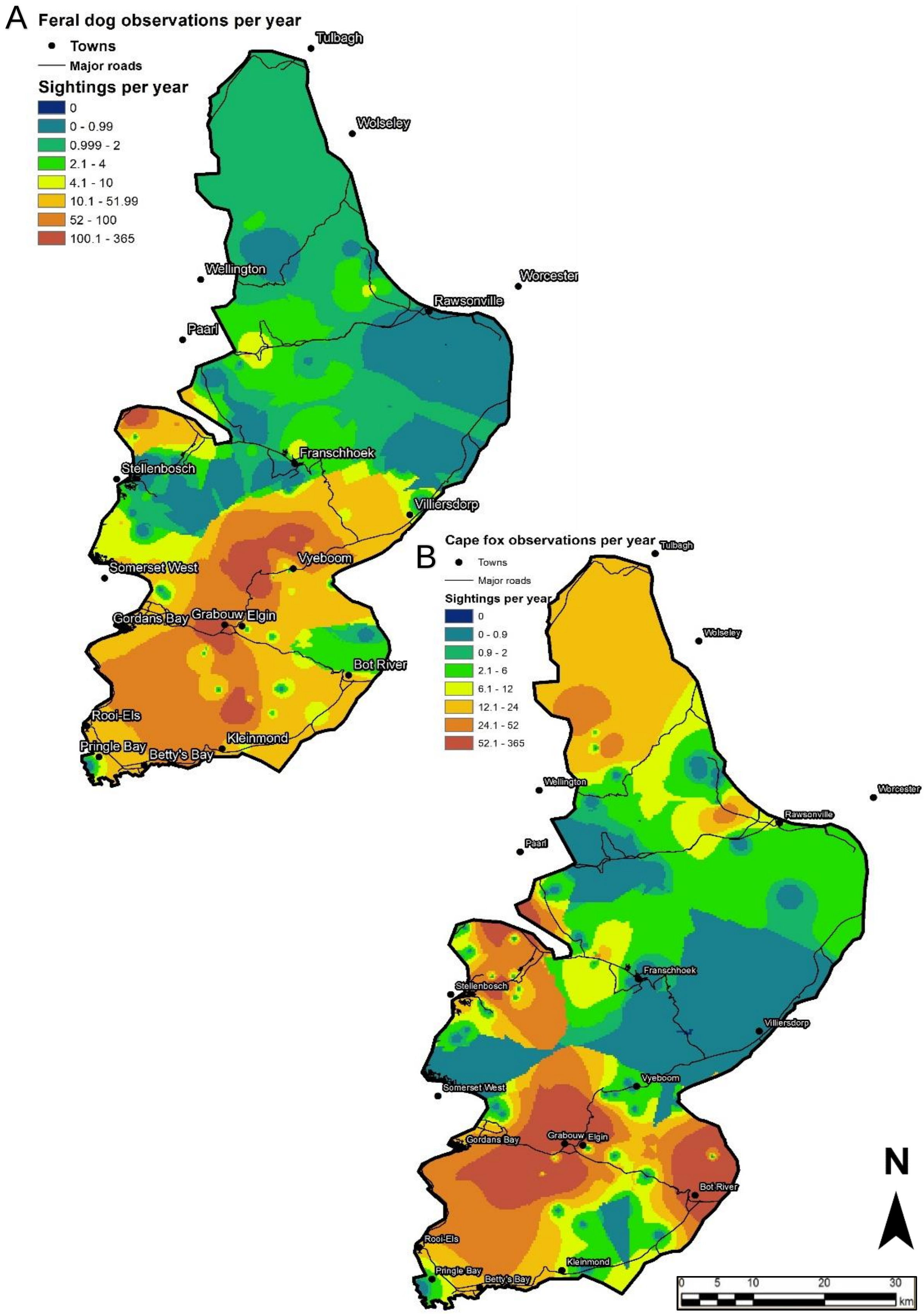


Figure 3.7: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Feral dog B: Cape fox).

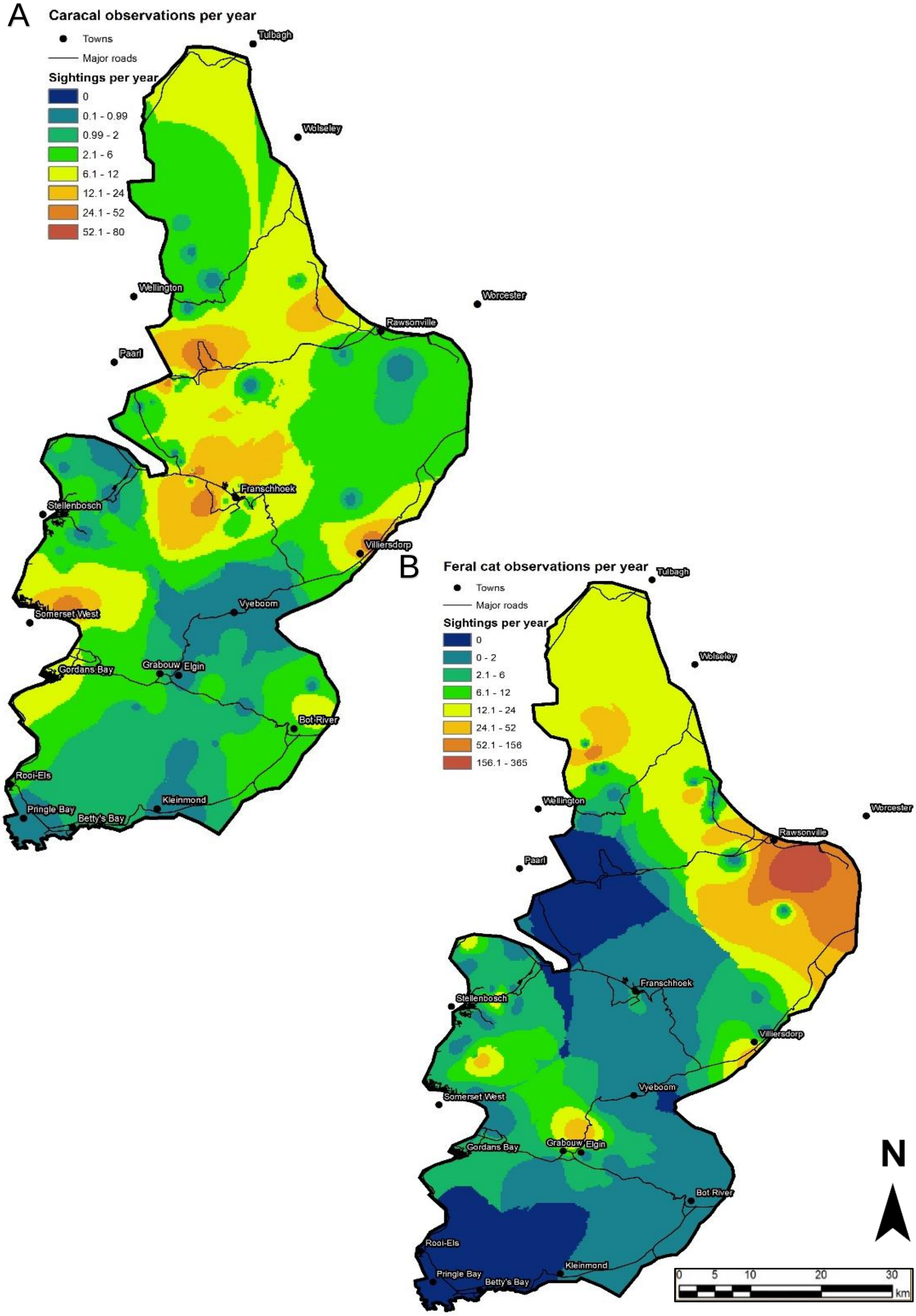


Figure 3.8: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Caracal B: Feral cat).

Grey rhebok were reported on a significantly higher percentage of farms in the KBR (61%) (Chi-squared (df=1)=6.66, $p<0.01$) than in the CWBR (28%) (Chi-squared (df=1)=6.66, $p<0.01$, Fisher's exact $p=0.01$). At the MZs level, rhebok were reported present on a significantly small percentage of farms in the West Hottentots-Holland MZ (17%) and Simonsberg MZ (10%) (Chi-squared (df=6)=15.07, $p=0.0197$) (Figure. 3.10A)

African striped weasels were reported present on significantly lower percentages in the Theewaterskloof Basin MZ (22%) and at significantly higher percentages of farms in the East Hawequas MZ (90%) (Chi-squared (df=6)=12.96, $p=0.044$), and at significantly higher frequencies in the East Hawequas MZ (49 per year) (LS means $F(6, 92)=2.876$, $p=0.01$) (Figure. 3.10B).

Feral pig were reported on a significantly lower percentage of farms in the KBR (0%) than the CWBR (17%) (Chi-squared (df=1)=5.17, $p=0.023$). Feral pig were reported present on a significantly greater percentage of farms in the West Hawequas (29%) (Wellington), Groenlandberg (18%), Theewaterskloof (11%) and West Hottentots-Holland MZs (9%) (Franschhoek) (Chi-squared (df=6)=13.85, $p=0.031$) (Figure. 3.12B).

Klipspringer were reported on a significantly higher percentage of farms in the Theewaterskloof Basin MZ (89%), and significantly less in the West Hottentots-Holland (17%) and Simonsberg MZs (20%) than the study area's other MZs ((Chi-squared (df=6)=15.07, $p=0.197$. Fisher's exact $p=0.02$) (Figure. 3.12B).

Leopard were reported on a significantly higher percentage of farms in the East Hawequas MZ (Slanghoek) and the Theewaterskloof Basin MZs (Villiersdorp) (LS means $F(96, 92)=2.511$, $p=0.03$) (Figure. 3.8B). The frequency of sightings was also significantly higher in East Hawequas MZ (Slanghoek) (less than once per year) that other MZs (Figure. 3.14A).

3.4.3 Distance to settlement

Wildcat (Welch test $p=0.03$), genet (Welch test $p=0.02$), honey badger (Welch test $p=0.05$), rhebok (Mann-Whitney U $p=0.03$), bat-eared fox (Mann-Whitney U $p=0.03$) and feral cat (Welch test $p<0.01$) were present on a significantly higher percentage of farms that were greater distances from settlements. These four species were less likely to be observed on a farm that was within 2.5km of a human settlement.

Hare (Spearman $r=0.29$, $p=<0.01$), large grey mongoose (Spearman $r=0.21$, $p=0.04$), leopard (Spearman's $r=0.29$, $p=<0.01$), duiker (Pearson's $r=0.22$, $p=0.03$) and klipspringer (Mann-Whitney U $p=0.05$) had significant positive correlations (Spearman's rank $r = 0.28$, $p=<0.01$) between their frequencies of sightings and distance to settlements.

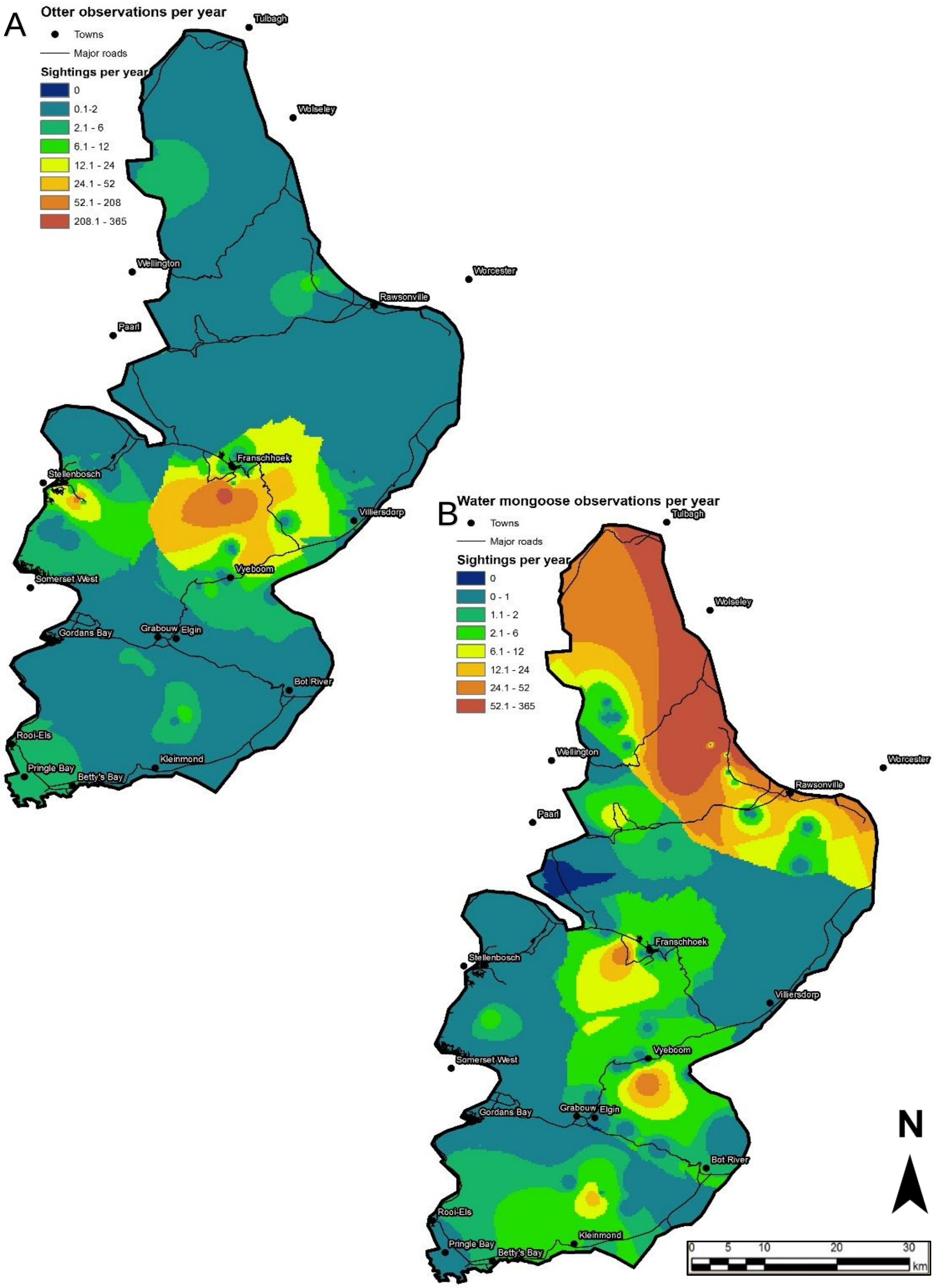


Figure 3.9: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Cape clawless otter, B: Water mongoose).

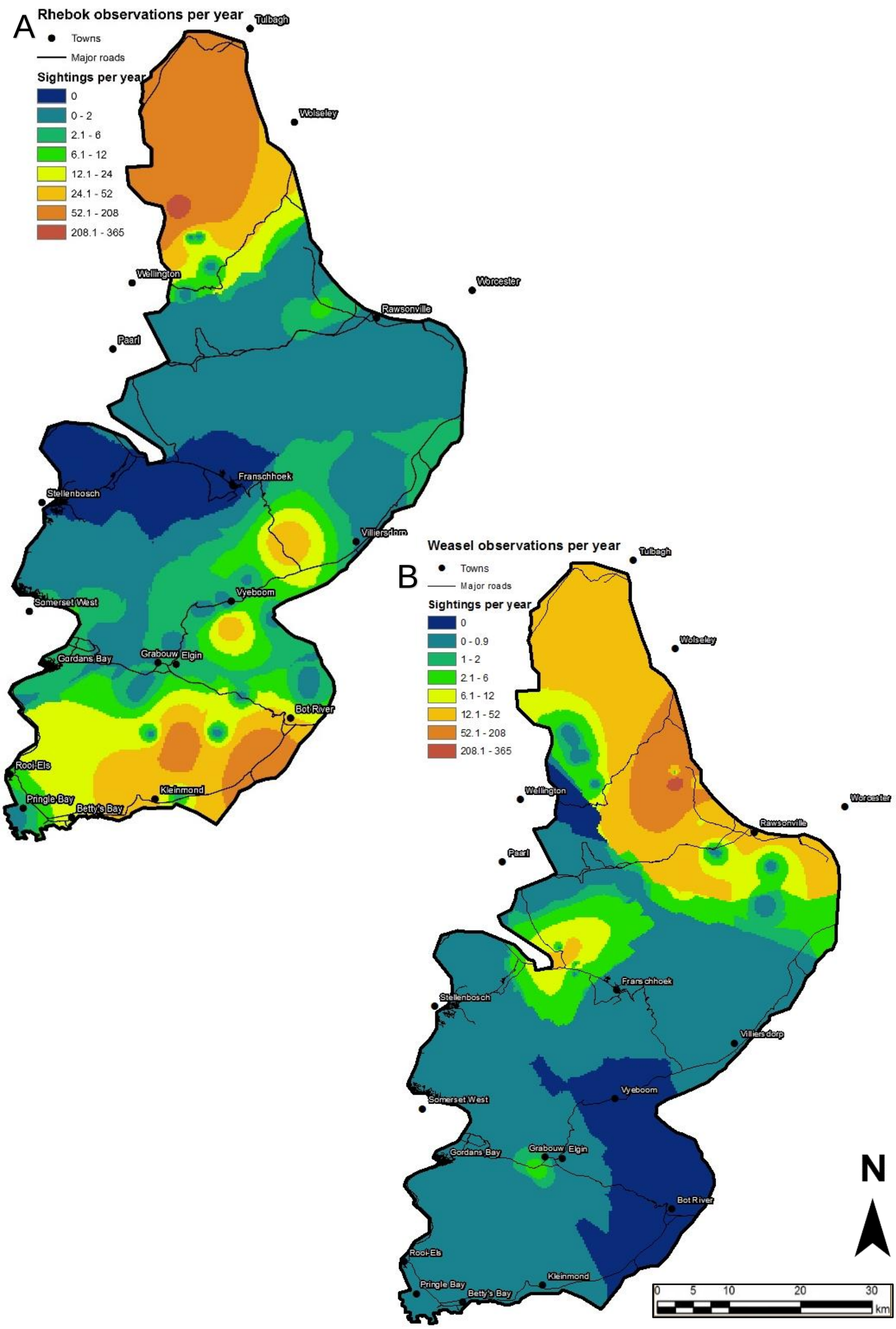


Figure 3.10: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Grey rhebok, B: African striped weasel).

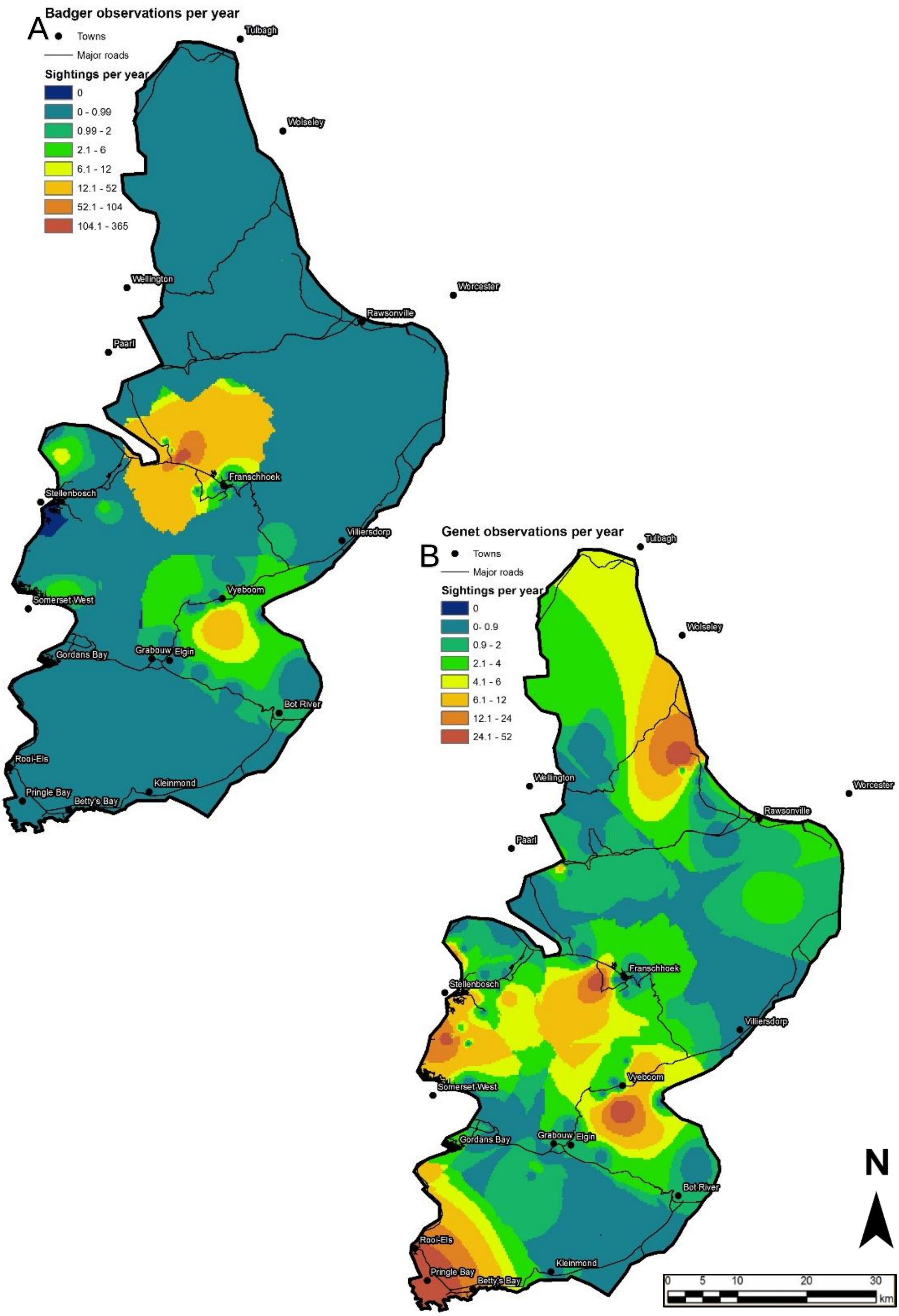


Figure 3.11: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Honey badger B: Genet spp.).

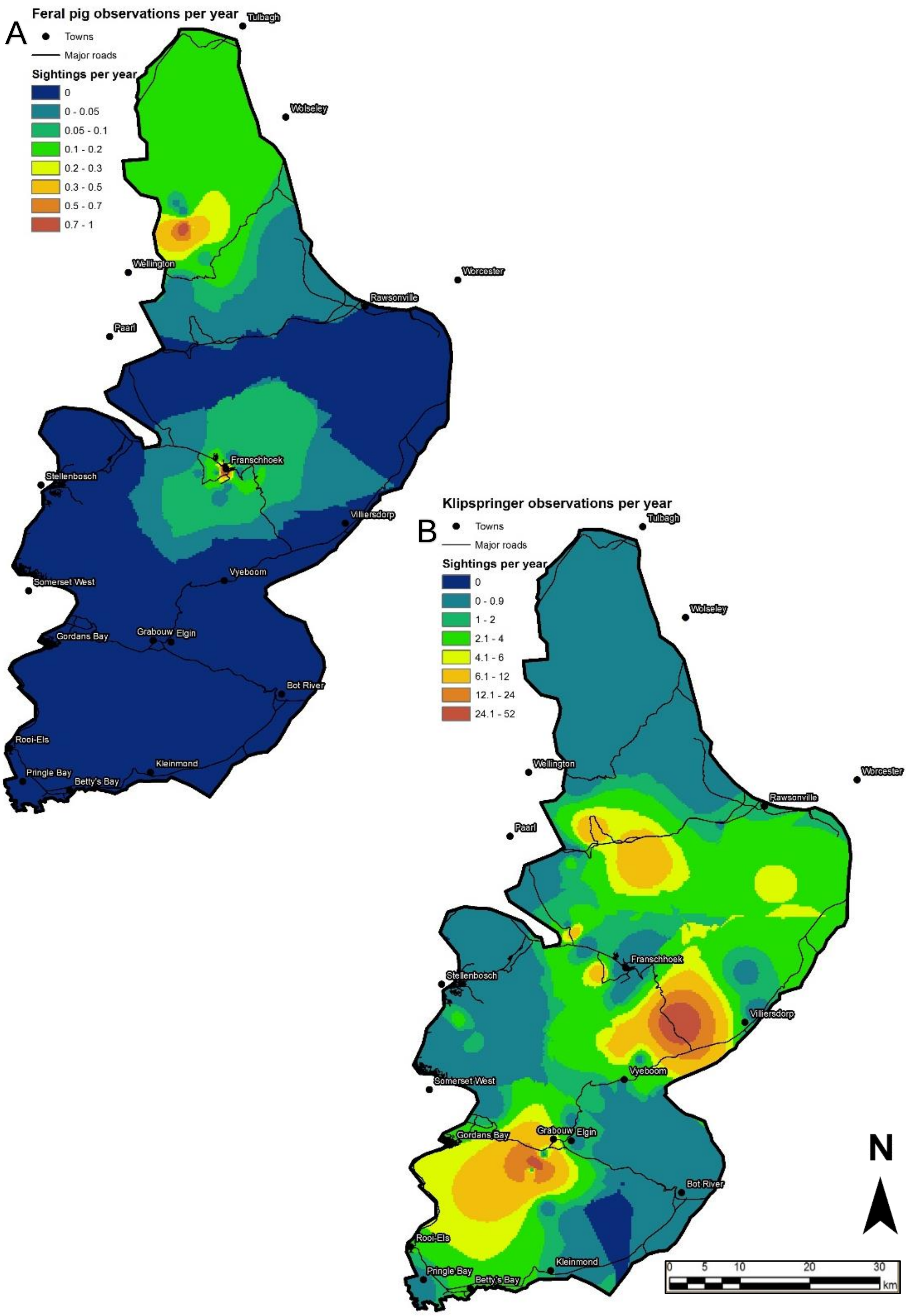


Figure 3.12: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Feral pig B: Klipspringer).

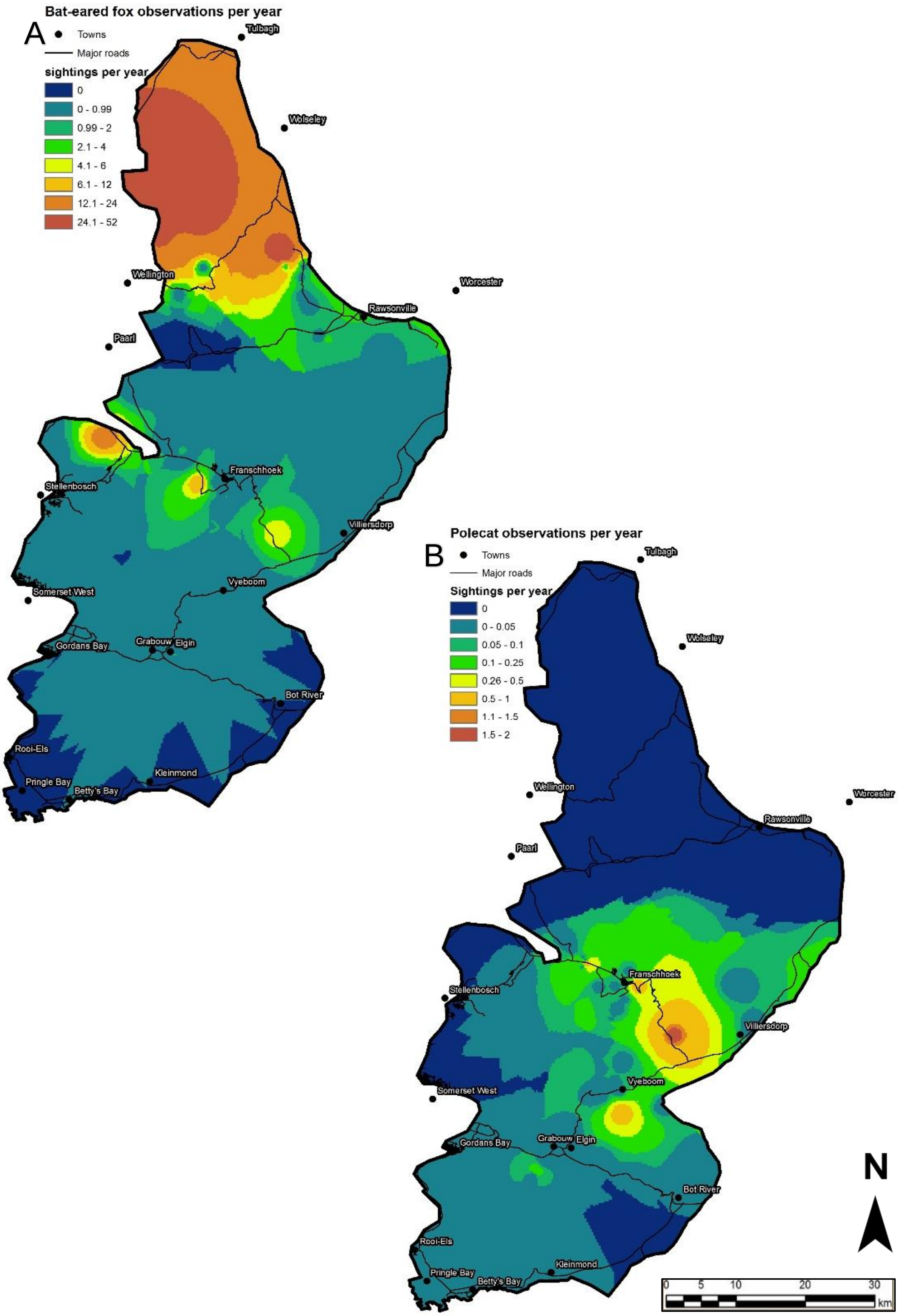


Figure 3.13: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Bat-eared fox B: Striped polecat).

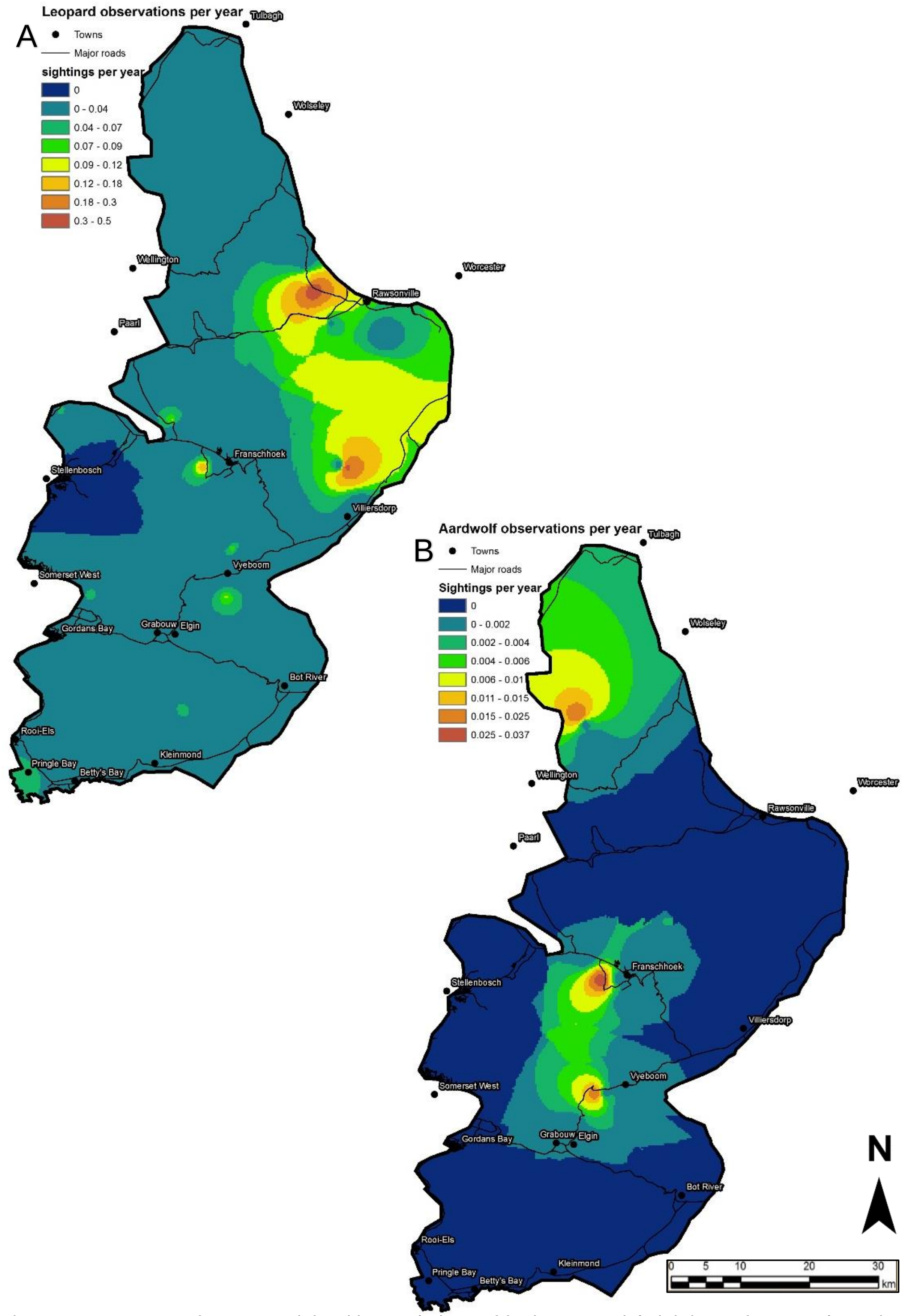


Figure 3.14: Inverse Distance Weighted interpolations of further potential sightings of mammal species per year throughout the study area surrounding the sample points (A: Leopard B: Aardwolf).

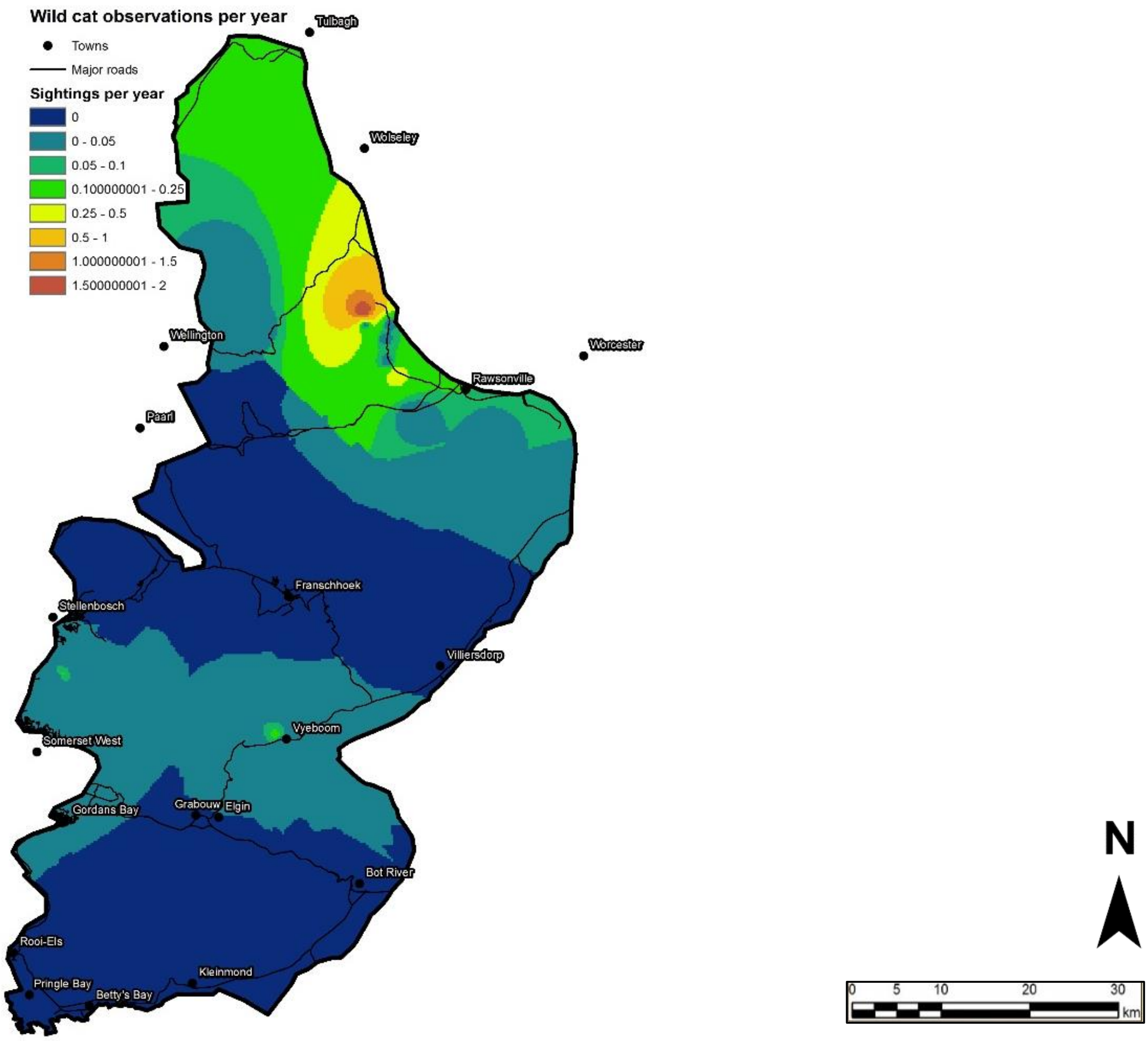


Figure 3.15: Inverse Distance Weighted interpolation of further potential sightings of mammal species per year throughout the study area surrounding the sample points (African wild cat).

3.4.4 Distance to major roads

Hare (Welch test p<0.01), rhebok (Welch test p<0.01), genet (Mann-Whitney U p=0.01) and feral pig (Mann-Whitney U p=0.03) were present on a significantly higher percentage of farms that were greater distances from major roads, occurring on farms further than 1.5km, 2.0km,1.75km and 2.5km from major roads respectively. Rock rabbits (Welch test p<0.01) were present on a significantly higher percentage of farms that were shorter distances from major roads. They were therefore less likely to be observed on a farm that was further than 1.75km of a major road. Hare (Spearman r=0.21, p=0.04), rhebok (Spearman r= 0.36, p<0.01) and genet (Spearman r=0.33, p<0.01) had significant positive correlations between their frequencies of sightings and distance to major roads.

3.4.5 Distance to protected area

Klipspringer (Mann-Whitney U p=0.05), genet (Mann-Whitney U p<0.01) and striped polecat (LS mean p=0.05) were present on a significantly higher percentage of farms that were shorter distances from PA boundaries. All three of these above-mentioned species were more likely observed within 1.5km of the boundary. Hares (Welch test p<0.01), African striped weasels (Mann-Whitney U p=0.02) and feral dogs (Mann-Whitney U p=0.01) were present on a significantly higher percentage of farms that were further distances from PA boundaries. Hares were therefore more likely to be observed further than 1.1km and weasels and feral dogs further than 1.2km of the PA boundary.

Genet (Spearman r= -0.23, p=0.02), klipspringer (Spearman r=-0.32. p<0.01), baboon (Spearman r= -0.26, p=0.02) and rock hyrax (Spearman r= -0.42, p<0.01) had significant negative correlations between their frequencies of sightings and distance to PA boundaries.

Percentage of natural land-use

Leopard (Mann-Whitney U p=0.05), rock hyrax (Mann-Whitney U p<0.05) and klipspringer (Mann-Whitney U p<0.01) were present on a significantly higher percentage of farms consisting of a greater percentage of natural land-use. Leopard, rock hyrax and klipspringer were more likely to be observed on farms with at least 45%, 50% and 55% natural land-use, respectively.

The frequencies that rock hyrax (Spearman r=0.34, p<0.01) and klipspringer (Spearman r=0.34, p<0.01) were sighted showed a significant, positive correlation to the percentage natural land-use. The frequencies that duikers (Pearson's r= -0.27, p<0.01) were sighted showed a significant, negative correlation to the percentage natural land-use.

3.4.6 Factors related to perceived changes in abundance

Caracals were the only species to show a consistently stable perceived population status, rather than increasing or decreasing (Chi-squared (df=18)=44.39, p<0.001; Fisher Exact (r x c) p<0.01 Fisher Exact (r x c) p<0.01).

Baboon populations were perceived to be decreasing significantly more (than those that remained stable or increasing) on farms closer to human settlements (Welch test F(2.0,34.9)=5.76, p<0.01) (Figure. 3.16). Rock hyraxes showed a reported significant relationship between population change and the distance from major roads (Welch test F(2.0,34.9)=5.76, p<0.01). Decreasing populations were reported on farms closer to major roads significantly more than populations that remained stable or increasing (Figure. 3.17).

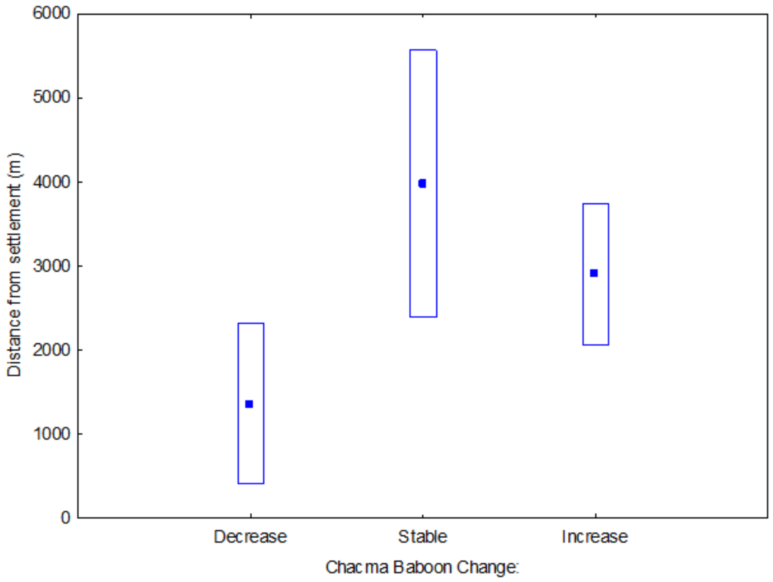


Figure 3.16: One-way ANOVA of distance from settlements (m) and the change to baboon populations. Vertical bars denote 0.95 confidence intervals.

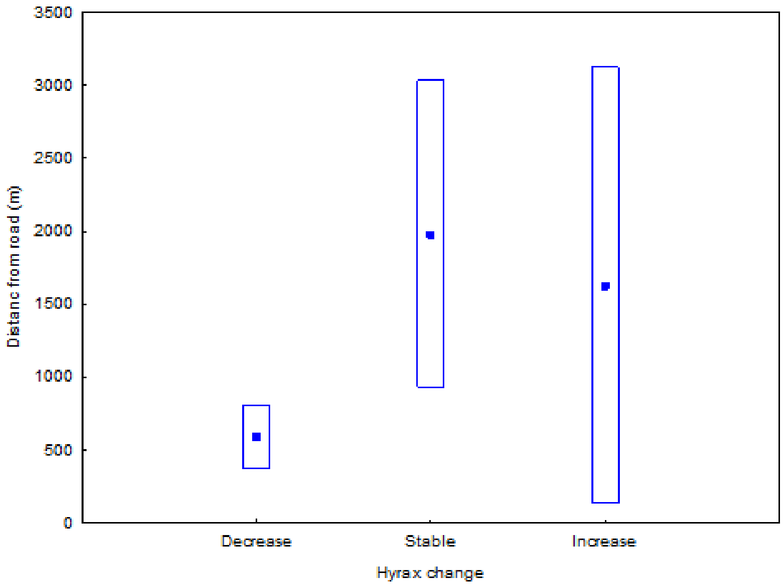


Figure 3.17: One-way ANOVA of distance from major roads (m) and the change to rock hyrax populations. Vertical bars denote 0.95 confidence intervals.

3.5 Discussion

A high diversity of medium-sized mammals is present on buffer agricultural properties in the BMC. This was demonstrated by the study detecting all the medium-sized mammal species that were previously documented in the BMC, having moved through or utilised agro-ecosystems (Wilkinson, personal communication, 2017). The abundance-distribution rank profile (Figure 3.2) illustrated typical, skewed patterns seen in many ecosystems, of a few common species and many scarce species (Verberk, 2011). Some bias may exist due to the variations in a species' behaviour, reflecting when and where they are most active, although this does then represent their distribution (Boakes et al., 2010). Interviewee bias to sighting more charismatic species is possible, but unlikely a significant concern as many species that are "less charismatic" were still detected at relatively high perceived rates (Boakes et al., 2010). The three most common species (chacma baboon, grey duiker and Cape grey mongoose) showed a pattern of visible unequal abundances. This large drop in abundances is typically seen in ecosystems where a disturbance prevents other, less adapted species from becoming abundant (Verberk, 2011). The anthropogenic effects of agro-ecosystems are likely the disturbances that act on these medium-sized mammals in the BMC. As other research has highlighted, baboons and duiker are highly adapted to agricultural land-uses (Dasmann & Mossman, 1962; Hoffman & O'Riain, 2012; Matthiae & Stearns, 1981). These three species are diurnal which also enhances their likelihood of being observed by farm labourers (Do Linh San & Cavallini, 2015; Hoffman & O'Riain, 2012; Nyhus et al., 2003; Stuart & Stuart, 2015).

The species that followed, displayed much more gradual levels of abundance. The scarcest species in the agricultural buffers (aardvark, aardwolf, Smith's red rock rabbit, African wildcat, leopard and striped polecat) showed similar abundances to one-another. All six of these species are typically reported as being nocturnal or crepuscular, which lowers their chances of being sighted during typical farm working hours (Apps, 2012; Do Linh San & Cavallini, 2015; Nyhus et al., 2003; Stuart & Stuart, 2015). Many scarce species in an ecosystem are often driven by more structurally complex ecosystems (Brown, 1984; Verberk, 2011). Changes to and inconsistencies in the abundance and distribution of those species that are more common in an ecosystem, could be indications of unnatural disturbances (Verberk, 2011).

3.5.1. Variation among species' abundances

Chacma baboons were the most common mammal in agro-ecosystems, without considering their large troop sizes (which can exceed 100 individuals) (Hoffman & O'Riain, 2012). Simonsberg MZ however, showed much lower baboon abundances, reinforcing the statement by many respondents that baboon populations have been decimated in this MZ (Wilkinson, personal communication, 2017). The current study confirms that baboons persist strongly in agricultural buffer zones within the BMC (Hoffman & O'Riain, 2012; Park, 2014). Human-wildlife conflict is a most probable driver for the population's decline on farms closer to human settlements (Kaplan et al., 2011). Baboons are major, expensive pests to plant-based agriculture and exhibit dangerous behavioural adaptions of raiding human waste and homes (Kaplan et al., 2011). Changes to baboon populations are either through their removal by humans or a behavioural change in the species to avoid settlements and seek out PAs, as a result of fear instilled by humans. The trend of higher baboon abundance on farms closer to PAs was also seen in this study.

Hares occurred at medium-high abundances in the BMC's agro-ecosystems but had a much more widespread presence in the KBR than the CWBR. Their relatively high perceived abundance may indicate their actual abundance is even greater as they are a predominantly nocturnal species which are less likely to be sighted during farm operation hours (Stuart & Stuart, 2015). Hares have been documented as roadkill in many studies and roads, as features, are responsible for many other negative impacts to natural habitats (D'Alton et al., 2002). These lower abundances were illustrated as low frequencies of sightings in Simonsberg MZ and in Stellenbosch, Franschhoek and Paarl, which are some of the oldest and highest populated towns in the BMC (Statistics South Africa, 2018; Stevens, 2003). This, with the trend of lower abundances near to major roads, alludes to features of human development as probable threats to hare species. D'Alton et al. (2002) noted a low occurrence of Cape hare in the nearby towns of Porterville, Swellendam and Bredasdorp. With two hare species inhabiting the BMC, their low abundance is concerning for both. The species were contrastingly more common on farms further from PAs. This may be due to the Cape hare's preference for grassland habitats and both species' dietary need for grass, which is typical of lowland areas in the Fynbos biome (Stuart & Stuart, 2015). The drivers from both PAs and human development leave the hare populations with more narrow available habitats. The species also have many natural predators in the BMC, such as caracal (whose populations are perceived as stable) (Boshoff et al., 1990, Palmer & Fairall, 1988). Introduced predators, such as domestic dogs and cats or feral pigs (which are also of medium-abundance) may also prey on hares (Hennig, 1998; Liberg, 1984).

Rock hyrax were another relatively common species to display some unequal abundances in the BMC. Their narrow abundance distributions on farms in the Groenlandberg MZ were also seen in the majority of the Kogelberg MZ's sample sites and in parts of the West Hottentots-Holland MZ. There was little indication of what may be driving this, but abundances of populations closer to roads showed a decline. The drivers for this decline may be wildlife-vehicle collisions, or more likely roads' role as vectors for pathogens, parasites and invasive species (Barry & Mundy, 1998; Parson et al., 2008; Wimberger et al., 2009). Reports of rock hyrax roadkill are scarce, but they are prone to various diseases and parasites and were reported as prey items to domestic dogs by many of the respondents and in Aticken et al.'s

(2009) study in Ethiopia. Respondents in the Groenlandberg MZ also reported domestic dogs being dumped on roadsides. Rock hyrax are preyed upon by many other native predators and therefore experience natural pressures on their population sizes (Avery et al., 2010; Klein & Cruz-Uribe, 1996; Palmer & Fairall, 1988; Wimberger et al., 2009). Human-wildlife conflict may also be a threat as they have been reported in this and other studies as a pest in orchards (Morhan & Cohen, 1987). Although rock hyrax were ranked at low preference to and frequency of being caught in snares, the Groenlandberg, Simonsberg and the West Hottentots-Holland MZs were marked as hotspots for snaring by Nieman et al.'s (2019) study.

Grysbok were generally a widespread species in the BMC but were also not present on many of the Groenlandberg MZ's farms. There is the potential for whatever is affecting the rock hyrax populations, to be negatively influencing grysbok populations as well. This may be due to the high incidence of snaring in the area and high preference and incidence of grysbok as targets within the BMC's agricultural buffers (Nieman et al., 2019).

Grey rhebok are another species that had low abundances in the Simonsberg MZ and the majority of the West Hottentots-Holland MZ and into the southern half of the Hawequas MZs. All three of these areas were noted as wire-snaring hotspots by Nieman et al. (2019) and rhebok are another desirable species that was occasionally reportedly snared. Their low abundances in these MZs contributed to the lower abundance in the CWBR when compared to the KBR. Aspects of roads and settlements are again probable disturbances that prevent grey rhebok from becoming more abundant (Verberk, 2011). Their habitat preference for flat plains and lower gradient slopes with grasses for shelter are typical habitats over which the BMC's agriculture and human settlements have been established and are a likely driver for so few sightings (Boshoff et al., 2001; Rouget et al., 2003; Taylor et al., 2017). Additionally, Taylor (2005) reported that ranges were smaller in size on steeper slopes, which may have lowered the chance of respondents sighting individuals. The species was however, recently upscaled from Least Concern (LC) to Near Vulnerable (NV) due to populations declining by 20% (Taylor et al., 2017). With their appearance on CapeNature's list of priority species for 2017 to 2021, the inconsistencies of abundances displayed in the current study is of great concern (Birss, 2017). Caracal could contribute to these lower grey rhebok abundances in the CWBR, as they are their preferred prey in the Karoo region (Palmer & Fairall, 1988).

Many of these results simply reflect natural differences in distributions and habitat preferences. For example, rock hyrax, klipspringer and leopard showed higher abundances on farms with more natural areas, and they generally prefer rocky outcrops and mountain slopes (Stuart & Stuart 2015). Natural areas offer these more ideally suited habitats. Typical farms in the BMC will however either clear rocky outcrops or avoid them and mountain slopes when designating locations for orchards and vineyards (Rouget et al., 2003). Those species that occur at higher abundances closer to PAs likely do so for similar reasons, such as genet requiring more dense vegetation for shelter, for example (Stuart & Stuart, 2015). The results showed that often the above-mentioned species are more abundant further from human settlements. This is attributed to their lower tolerance for human disturbances and that these are parallel factors.

The East Hawequas MZ had higher abundances of multiple medium-sized mammalian species than elsewhere in the BMC. This might be due to its semi-arid climate and partial Nama-karoo vegetation and ecotones, which is a biome known to support more mammals than more humid climates (Greef, 1991; Lloyd, 2000; Mann et al., 2019; Rutherford et al., 2002).

Simonsberg MZ held the lowest abundances for many species in the BMC. Baboons, hares, large grey mongoose, klipspringer and grey rhebok were recorded at their lowest frequencies and on the fewest farms in the Simonsberg MZ. The isolation of this MZ from the rest of the core PAs in the BMC is the potential driver for these low species abundances (Rogan & Lacher Jr., 2018). As an isolated habitat patch, Simonsberg MZ's core surface area is proportionately smaller than its surrounding agricultural buffers (Pardini et al., 2018). This would leave its medium-sized mammal species more vulnerable to edge effects and anthropogenic disturbances (Elias, 2014; Rogan & Lacher Jr., 2018). Those-mentioned species with lower perceived abundances in the Simonsberg MZ, may be experiencing the isolated patch as population sinks (Newmark, 2008), although, the majority of the same species (except baboons) were also scarcely abundant in most of the nearby West Hottentots-Holland MZ. Both above-mentioned MZs' farms buffer between PAs and some of the most populated towns in the BMC (Statistics South Africa, 2018). Wire-snaring was mentioned by respondents throughout the study area and in the Simonsberg and West Hottentots-Holland MZs. Nieman et al.'s (2019) study indicated that the Simonsberg MZ and an extended area toward Stellenbosch were hotspots for wire snaring. This, with other potential threats is likely a driver from human development for this significant difference in some species lower perceived abundances (Kahler & Gore, 2015; Newmark, 2008). Other aspects of human developments, such as habitat loss (Lombard et al., 1997), exposure to fire ignition sources (Rebelo, 1992), invasive species (Botha, 1989), introduced toxins (Serieys et al., 2019), human-wildlife conflict (Martins & Martins, 2006), higher densities of roads and traffic (Newmark, 2008) and various pollution sources (Newmark, 2008) are more likely to act on greater scales and intensities near these towns (Pardini et al., 2018).

3.5.2 Detected threats

Birss (2017) had indicated feral pig populations were distributed in the West Hawequas MZ and Botriver (Groenlandberg MZ). The current study concurred that populations were the most abundant in the West Hawequas MZ and had a lower, but significant abundance in the Groenlandberg MZ. However, a wider distribution of feral pig populations in the Theewaterskloof Basin MZ and, as Botha described in 1989, in Franschhoek, was shown by the results of the current study. Feral pigs can cause damage to lowland fynbos and renosterveld vegetation which may alter these habitats' ability to sustain medium-sized mammal populations (Birss, 2017; Picker & Griffiths, 2011). Feral pigs are described as aggressive and have been recorded preying on lambs and other smaller mammals (Botha, 1989; Picker & Griffiths, 2011). Although not visible in other populations, feral pigs in the Groenlandberg MZ could therefore, have contributed to the lower abundances of rock hyrax and grysbok seen in this MZ.

Prior to this study, feral and free-roaming domestic dog population presences were only anecdotally reported (and more recently) by agricultural stakeholders (Wilkinson, personal communication, 2018). Feral dogs are an invasive predator detected at medium abundances evenly throughout the BMC agro-ecosystems. The species showed greater abundances further from PAs, which could indicate their dependency on human development for survival, as many studies have shown (Silva-Rodrígue & Sieving, 2012; Vanak & Gompper, 2010; Woodroffe & Donnelly, 2011). However, they potentially have many impacts on natural habitats and do affect a high diversity of native medium-sized mammals as shown in other studies (Young et al., 2011). Duiker, grysbok, klipspringer, rhebok, baboon, porcupine, rock hyrax, hares, mongoose, weasel, polecat, genets, eland, livestock (poultry, calves), rodents, birds and reptiles were animals reported in this study to be disturbed, chased, caught and/or killed by free-roaming dogs in the BMC. As the perceived third most abundant predator in the BMC it is clear how they can have such severe impacts. Feral dogs can be large and often hunt in packs in other ecosystems (Atickem et al., 2009). Packs were noted in the BMC's agro-ecosystems by stakeholders during this study. A respondent in the Kogelberg MZ also reported a pack of feral dogs destroying an entire family group of 12 bat-eared foxes in one day. These are the combined effects of free-roaming hunting and feral dogs, but accurate sources and effects of each could not be determined from the current study.

Poaching with the use of wire snares was frequently mentioned, and ways in which many respondents had observed some mammals was as a carcass in a wire snare. This method of poaching was recently described by Nieman et al. (2019) as prolific throughout agricultural buffers zones in the BMC, within the study area. The authors reported the majority of the current study's species being affected by snares, including leopard. Human-wildlife conflict was also mentioned during the current study, and various farmers took measures into their own hands to eradicate species such as baboon, porcupine, feral dogs and feral pigs. CapeNature has a permit system for the regulation of hunting of many of the study species on private lands, but without adequate enforcement this may strongly influence medium-sized mammal compositions in the landscape (Department of Environmental Affairs, 2010). Fences are another component of many human developments, (particularly in South Africa) that were mentioned as reasons for changes in medium-sized mammal compositions in buffer zones. Game fences and security fences may restrict what mammals can access a habitat, without that habitat changing in land-cover (Boone & Hobbs, 2004; Newmark, 2008). Many of the species were also sighted as roadkill from vehicle-wildlife collisions.

3.5.3 Implications for leopard

The low abundance derived from the reported leopard sightings in the BMC's was expected, based on their natural secretive, nocturnal habits and low densities in the Fynbos biome (Martins & Martins, 2006). Most of the recorded prey species' high diversity was perceived at greater abundances than leopard (Mann et al., 2019). This reduces the concern over prey availability (Martins et al., 2010). Leopard had greater abundances on farms with similar compositions of larger percentages of natural area, as klipspringer and rock hyrax did. This overlap supports Fröhlich's (2011) statement that leopard favour the same ranges where their most common prey species (klipspringer and rock hyrax) occur (Mann et al., 2019). Many of their main prey species (duiker, hare, rock hyrax, porcupine and grysbok) have high to medium abundances on farms. Lower rhebok abundances in some parts of the BMC, could be the reason why they are not as common a prey species of leopard in other Western Cape ranges (Hayward et al., 2006). The presence of feral pig in some MZs may benefit leopard, as they have been recorded as prey items in multiple studies, both in the BMC and savannas (Botha, 1989; Norton et al., 1986).

The noted lower abundances of some main prey species in areas such as the Groenlandberg, Simonsberg and West Hottentots-Holland MZs are not too great of a concern for leopard. This is because of the high abundances of other prey options available on agricultural buffers and the species' ability to move into other areas for better prey access (Rogan & Lacher Jr., 2018). What is concerning, however, is whether leopard are also at risk to whatever has driven the scarcity of certain species in these areas. There are reservoirs of prey-species in agri-ecosystems, therefore prey loss is not as great of a concern when compared to the mortality risks that leopard encounter on these properties when hunting. The large home range sizes of leopard in the Cape can intensify the threats of habitat isolation (for e.g.: Simonsberg MZ)

(Martins & Martins, 2006; Rogan & Lacher Jr., 2018). Threats reported by respondents in this area include wildlife-vehicle collisions, poaching (wire snares, packs of dogs and porcupine traps), retaliatory killing during human-wildlife conflict, land-cover change, fences (which limit access to habitats), fires, spread of invasive vegetation and free-roaming dogs.

The feral dogs detected pose three major threats to leopard in the BMC: as competitors over habitat and prey resources, as pathogen and parasite vectors and as predators (Butler & du Toit, 2002; Butler, du Toit & Bingham, 2003). The current study was able to detect that feral dogs have an impact on at least eight species commonly consumed by leopard (Hayward et al., 2006; Mann et al., 2019; Martins et al., 2010). The low abundances of many of these species increases feral dogs' negative ecosystem effects and as competitors to leopard (Butler & du Toit, 2002; Less et al., 2016). This pressure of competition with leopards is heightened by the stable abundances of caracal and high diversity of meso-carnivores present in the BMC (Norton & Lawson, 1985).

A pack of feral dogs could easily hunt one of the small-sized local leopard (Atickem et al., 2009; Stein et al., 2016). Another anthropogenically-driven sources of mortality in the BMC in these scarce leopard populations, can have dire impacts (Martins & Martins, 2006). Many studies in Africa and Asia have however, recorded the predation of feral dogs by leopard (Athreya et al., 2014; Butler et al., 2014; Butler et al., 2003). As with feral pigs, feral dogs may become a prey source for local leopard if natural prey species are depleted (Botha, 1989; Butler et al., 2014; Norton et al., 1986), although this may be unlikely due to the smaller body size of Cape leopard in general (Athreya et al., 2014; Butler et al., 2003; Stein et al., 2016). The hunting of feral dogs by leopard would enhance the risk of them contracting diseases such as canine distemper and rabies (Butler et al., 2003). The current study has shown that there is a sufficient abundance and diversity of leopards medium-sized mammalian prey species across agro-ecosystems. However, of concern is the high perceived abundance of feral dogs in the study areas landscape that pose a much greater, overall threat to leopard and medium-sized mammals survival.

As mentioned above, wire snaring is the method of hunting that drives the highest concern to all medium-size mammals in the BMC because of their widespread reported occurrence across the BMC, how easily and affordable they are to set in large numbers, their indiscriminate selection of species captured and the difficulty in controlling/monitoring their use (Lindsey et al., 2011; Noss, 1998). Overtime the negative impacts of wire-snaring on mammals has been recognised in more PAs across Africa, and in the last year Nieman et al. (2019) highlighted its widespread incidence in the BMC (Gandiwa et al., 2013; Kaschula & Shackleton, 2009; Lindsey et al., 2013). The majority of the study species are threatened by snaring because of its unselective impact on what species are trapped, thus all mentioned species populations of concern from this study would be negatively affected by snaring (Becker et al., 2012; Lindsey et al., 2011). Nieman et al. (2019) further found the most sought-after mammals to hunt and those that were most frequently captured are many of the primary prey species of leopard in the BMC (Mann et al., 2019; Martins et al., 2011). Snares therefore threaten both leopard and their prey availability.

3. 6 Conclusions and future recommendations

3.6.1 Conclusions

The high diversity of mammals in the BMC is reflected in its buffer agro-ecosystems, where natural variations among medium-sized mammal abundances exist. These buffer zones contain many habitat disturbances that are generated by anthropogenic effects. These disturbances limit or enable the persistence of each species. The CWBR contains more variations in mammal abundances and is potentially exposed to more threats than the KBR. The greater exposure of edge-habitat within buffer zones, to human development and higher densities of human populations, enhances the negative impacts on multiple mammal abundances. Examples of such edge-habitats that appear to reflect negative impacts are those in buffer zones that border Somerset West, Stellenbosch, Franschhoek and Paarl (Statistics South Africa, 2018). Simonsberg MZ was most concerning during this study, because of its lower medium-sized mammal abundances, isolation and high exposure to developing towns.

Native mammal populations that raised some conservation concerns and may be at risk in the future, were the hare spp. and grey rhebok. Buffer habitats were conducive to abundances of invasive domestic mammal species. Feral pigs appear to be more widely distributed in the BMC than previously acknowledged. Feral dogs are of concerningly high and extensive abundance in this agro-ecosystem. Feral dogs are a documented threat to a great variety of naturally occurring mammals and other fauna as well as domestic animals. Leopard should persist with current trends in medium-sized mammalian prey abundances. The synergy of direct threats from human development that leopard are exposed to in buffer zones is likely to be a greater threat in the BMC than prey depletion. The threats that were detected in this study during sampling included poaching (wire snares, packs of dogs and porcupine traps), retaliatory killing during human-wildlife conflict, land-cover change, fences, wildlife-vehicle collisions, fires, functional habitat loss (corridors and lowland fynbos), spread of invasive vegetation and free-roaming feral dogs.

Buffer zones are adequate in supporting medium-sized mammals. It is important for properties in buffer zones to contain some natural, diverse habitat types. Renosterveld, lowland fynbos and grassland are particularly important habitat types for medium-sized mammal diversity. Natural habitats, corridors between core areas and a high diversity of species offer some resilience for leopard prey species in agricultural buffer zones.

3.6.2 Recommendations

Functions of the UNESCO Biosphere Reserves that must continue being implemented are: constant monitoring of both human developments and natural processes, and collaboration with researchers to study the species of concern highlighted in this study (Batisse, 1982; Ishwaran et al., 2008). Monitoring guidelines are implemented in most of the CapeNature reserves within the BMC (Birss, 2017). It is recommended that these guidelines be extended into the surrounding buffer agricultural areas (Cooper et al., 2007). A means of expanding these guidelines may be through concerted coordinated efforts of local registered conservancies which many of the study farms are a part of. Guidelines would need to be altered into a format that is straight forward to follow for agricultural property owners and stakeholders. These programmes could follow similar structures as citizen science (Cooper et al., 2007). It would be most beneficial to keep stakeholders on private properties involved in the monitoring systems, and further management decision making, because of their valuable LEK, constant presence and responsibility for these properties (Folke, 2004).

Mammal populations for which urgent monitoring is recommended, include: grey rhebok and both scrub and Cape hare. CapeNature has already listed that priority actions for grey rhebok are to collect distribution and population data between 2017 and 2021 (Birss, 2017). The biggest priority is to monitor their localized distributions, population sizes, how they are threatened and determine their conservation status within their Western Cape ranges (Stenkewitz et al., 2010). As D'Alton et al. (2002) suggested, the conservation status of hares may need to be evaluated on a regional scale in the Western Cape's PAs and buffer habitats. It is recommended that this be carried out throughout the BMC. Both scrub and Cape hare require individual species monitoring and conservation assessments. All MZs need comparative monitoring plans. Those closer to larger populated towns and/or isolated habitats need priority assessments. These towns include Stellenbosch, Somerset West, Franschhoek, Paarl and Grabouw (Statistics South Africa, 2018).

Feral dog research, monitoring and the development of a management plan is a top priority. It is important that the public and stakeholders are better informed and help conceptualise solutions instead of taking matters completely into their own hands (Agri SA, 2017; Massara

& Chiarello, 2012). This issue should be better documented and prioritised by CapeNature and other leading conservation authorities. To implement management strategies on free-roaming dogs, one needs to determine sources, drivers and the impacts these dogs are having. Thereafter, measures can be put in place to prevent free roaming dogs entering these habitats and spreading. Fast detection and removal of individuals is vital (Lessa et al. 2016). Due to the ethical conflicts regarding domestic dogs, further research is required into what other damages these populations are having on the ecosystem, to humans and their quality of life and health (Lumney et al., 2011).

Many aspects of landscape ecology need assessment. A recommended priority is that a larger scale investigation into fragmentation among PAs throughout the Western Cape and the functional connections between them be undertaken (Pardini et al., 2018). The thresholds of each mammal species to habitat isolation and fragmentation needs to be determined (Pardini et al., 2018; Rogan & Lacher Jr., 2018). As much habitat diversity and functional connection as possible needs to be sustained (Fattebert et al., 2013; Karanth & Chellam, 2009). Critical mammalian habitats (functional corridors, grassland, lowland fynbos and renosterveld) need to be identified in the landscape, both on private properties and PAs (von Hase et al., 2010). These and mammalian access to them, then need to be formally protected from human development. This likely falls under CapeNature's management and would benefit from successful enforcement of the National Environmental Management: Protected Areas Act (NEM:PAA, 2003) (Pool-Stanvliet et al., 2017). Of the 99 sampled properties 14 fall under either CapeNature's or the World Wide Fund for Nature's (WWF) conservation stewardship programmes (see https://www.capenature.co.za/care-for-nature/stewardship/ and https://www.wwf.org.za/our_work/initiatives/conservation_champions.cfm, respectively). It would be beneficial if more properties could join the programmes, especially those housing the above-mentioned habitats. This is another means to connect with stakeholders and implement all the above-mentioned monitoring programmes and actions.

3.7 Reference list

Agri SA. (2017). Illegal hunting with dogs – guidelines. www.agrisa.co.za/

Altrichter, M. & Boaglio, G.I. (2004). Distribution and relative abundance of peccaries in the Argentine Chaco: Associations with human factors. *Biological Conservation*, 116, 217-225.

Anadón, J.D., Gimenez, A. & Ballestar, R. (2010). Linking local ecological knowledge and habitat modelling to predict absolute species abundance on large scales. *Biodiversity Conservation*, 19, 1443-1454.

Anadón, J.D., Giménez, A., Ballestar, R., Pérez, I. (2009). Evaluation of local ecological knowledge as a method for collecting extensive data on animal abundance. *Conservation Biology*, 23(3), 617-625.

Athreya, V., Odden, M., Linnell, J.D.C., Krishnaswamy, J. & Karanth, K.U. (2014). A cat among the dogs: leopard Panthera pardus diet in a human-dominated landscape in western Maharashtra, India. *Oryx,* 1-7.

Atickem, A., Bekele, A. & Williams, S.D. (2009). Competition between domestic dogs and Ethiopian wolf (*Canis simensis*) in the Bale Mountains National Park, Ethiopia. *African Journal of Ecology*, 48, 401-407.

Avery, G. Robertson, A.S., Palmer, N.G. & Prins, A.J. (2010). Prey of giant eagle owls in the De Hoop Nature Reserve, Cape Province, and some observations on hunting strategy. *Ostrich*, 56(1-3), 117-122.

Azzuro, E., Moschella, P. & Maynou, F. (2011). Tracking signals of change in Mediterranean fish diversity based on local ecological knowledge. *PLOS ONE*, 6(9), 1-8.

Balme, G.A., Lindsey, P.A., Swanepoel, L.H. & Hunter, L.T.B. (2014). Failure of research to address the range wide conservation needs of large carnivores: Leopards in South Africa as a case study. *Conservation Letters*, 7(1), 3-14.

Barry, R.E. & Mundy, P.J. (1998). Population dynamics of two species of hyraxes in the Matobo National Park, Zimbabwe. *African Journal of Ecology*, 36, 221-233.

Batisse, M. (1982). The biosphere reserve: A tool for environmental conservation management. *Environmental Conservation*, 9(2), 101-111.

Becker, M., McRobb, R., Watson, F., Droge, E., Kanyembo, B., Murdoch, J., et al. (2012). Evaluating wire-snare poaching trends and the impacts of by-catch on elephants and large carnivores. *Biological Conservation*, 158, 26–36.

Begg, K.S. (2001). *Report on the conflict between beekeepers and honey badgers Mellivora capensis, with reference to their conservation status and distribution in South Africa.* Published to www.honeybadger.com (Unpublished report for the Endangered Wildlife Trust, Johannesburg).

Bencin, H., Kioko, J. & Kiffner, C. (2016). Local people's perceptions of wildlife species in two distinct landscapes of Northern Tanzania. *Journal of Nature Conservation*, 34, 82-92.

Bernard, H.R., Killworth, P., Kronenfeld, D. & Sailer, L. (1984). The problem of informant accuracy: The validity of retrospective data. *Annual Review of Anthropology,* 13, 495–517.

Birss, C. (2017). Mammals. In: Turner, A.A. (Ed.), *Western Cape Province State of Biodiversity 2017* (pp. 191-230). Stellenbosch: CapeNature Scientific Services.

Boakes, E.H., McGowan, P.J.K., Fuller, R.A., Chang-qing, D., Clark, N.E., O'Connoer, K. et al. (2010). Distorted views of biodiversity: spatial and temporal bias in species occurrence data. *PLOS Biology*, 8(6), 1-11.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2001). A pragmatic approach to estimating the distributions and spatial requirements of the medium- to large-sized mammals in the Cape Floristic Region, South Africa. *Diversity and Distributions*, 7, 29-43.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2002). Estimated spatial requirements of the medium- to large-sized mammals according to broad habitat units in the Cape Floristic Region, South Africa. *African Journal of Range and Forage Science*, 19(1), 29-44.

Boshoff, A.F., Palmer, N.G. & Avery, G. (1990). Regional variation in the diet of martial eagles in the Cape Province, South Africa. *South African journal of Wildlife Research*, 20(2), 57-68.

Braczkowski, A., Watson, L., Coulson, D., Lucas, J., Peiser, B. & Rossi, M. (2012). The diet of caracal, *Caracal caracal*, in two areas of the southern Cape, South Africa as determined by scat analysis. *South African Journal of Wildlife Research*, 42(2), 111-116.

Brashares, J. S., Golden, C. D., Weinbaum, K. Z., Barrett, C. B., & Okello, G. V. (2011). Economic and geographic drivers of wildlife consumption in rural Africa. *Proceedings of National Academy of Sciences*, 108(34), 13931-13936.

Butler, J.R.A. & du Toit, J.T. (2002). Diet of free-ranging domestic dogs (*Canis familiaris*) in rural Zimbabwe: Implications for wild scavengers on the periphery of wildlife reserves. *Animal Conservation*, 5, 29-37.

Butler, J.R.A., du Toit, J.T. & Bingham, J. (2003). Free-ranging domestic dogs (Canis familiaris) as predators and prey in rural Zimbabwe: Threats of competition to large wild carnivores. *Biological Conservation*, 115, 369-378.

Butler, J.R.A., Linnell, J.D.C., Morrant, D., Athreya, V., Lescureux, N., McKeown, A. (2014). Dog eat dog, cat eat dog: Social-ecological dimensions of dog predation by wild carnivores. In: M.E. Gompper, (Ed.), Free-ranging dogs and wildlife conservation (pp. 117–143). Oxford: Oxford University Press.

Burton, A.C., Sam, M.K., Kpelle, D.G., Balangtaa, c., Buedi, E.B. & Brushares, J.S. (2001). Evaluating persistence and its predictors in a West African carnivore community. *Biological Conservation*, 144, 2344-2363.

Caro, T.M. (2003). Umbrella species: Critique and lessons from East Africa. *Animal Conservation*, 6, 171-181.

Ĉervinka, J., Šálek, M., Padyšáková, E. & Šmilauer, P. (2012). The effects of local and landscape-scale habitat characteristics and prey availability on corridor use by carnivores: A comparison of two contrasting farmlands. *Journal of Nature Conservation*, 21, 105-113.

Child, M.F., Rowe-Rowe D., Birss, C., Wilson, B., Palmer, G., Stuart, C. et al. (2011). Conservation assessment of *Poecilogale albinucha*. In M.F. Child, L. Roxburgh, E. Do Linh San, D. Raimondo & H.T. Davies-Mostert (Eds.), *The Red List of Mammals of South Africa, Swaziland and Lesotho.* South Africa: South African National Biodiversity Institute and Endangered Wildlife Trust.

Clark, W.R. & Reeder, K.F. (2005). Continuous Conservation Reserve Program: Factors influencing the value of agricultural buffers to wildlife conservation. *Fish and Wildlife Benefits of Farm Bill Programs*, 93-113.

Cooper, C. B., Dickinson, J., Phillips, T. & Bonney, R. (2007). Citizen science as a tool for conservation in residential ecosystems. *Ecology and Society* 12(2), 11.

D'Alton, M.J., Kryger, U. & Suchentrunk, F. (2002). Possible range reduction of Cape hare (*Lepus capensis*) in the Overberg region of the Cape Province. *Research Institute of Wildlife Ecology, University of Veterinary Medicine*, Vienna, 59-61.

Dasmann, R.F. & Mossman, A.S. (1962). Abundance and populations structure of wild ungulates in some areas of Southern Rhodesia. *The Journal of Wildlife Management*, 26(3), 262-268.

Department of Environmental Affairs (2010). National environmental management: Biodiversity Act, 2004 (Act No. 10 of 2004). *Draft norms and standards for the management of damage-causing animals in South Africa*. Government Gazette 33806.

Driscoll, D.L. Appiah-Yeboah, A., Salib, P. & Rupert, D.J. (2007). Merging qualitative and quantitative data in mixed methods research: How to and why not. *Ecological and Environmental Anthropology*, 3(1), 19-28.

Elias, S.A. (2014). Rise of human influence on the world's biota. *Reference Module in Earth Systems and Environmental Science.* Amsterdam: Elsevier

Fairbanks, D.H.K., Hughes, C.J. & Turpie, J.K. (2003). Potential impact of viticulture expansion on habitat types in the Cape Floristic Region, South Africa. *Biodiversity and Conservation*, 13, 1075-1100.

Fehlmann, G., O'Riain, J.M., Kerr-Smith, C., Hailes, S., Luckman, A., Shepard, E.L.C. et al. (2017). Extreme behavioral shifts by baboons exploiting risky resource-rich, human-modified environments. *Scientific Reports*, 1- 8.

Fattebert, J., Dickerson, T., Balme, G., Slotow, R. & Hunter, L. (2013). Long-distance natal dispersal in leopard reveals potential for a three-country metapopulation. *South African Journal of Wildlife Research*, 43(1), 61-67.

Folke, C. (2004). Traditional knowledge in social–ecological systems. *Ecology and Society,* 9(3), 7.

Fröhlich, M. (2011). Studying the foraging ecology of leopards (*Panthera pardus*) using activity and location data: An exploratory attempt. Unpublished master's thesis, Humboldt University of Berlin.

Gandiwa, E., Heitkönig, I.M.A., Lokhorst, A.M., Prins, H.H.T., & Leeuwis, C. (2013). Illegal hunting and law enforcement during a period of economic decline in Zimbabwe: A case study of northern Gonarezhou National Park and adjacent areas. *Journal for Nature Conservation*, 21, 133-142.

Gastwirth, J.L., Gel, Y.R. & Miao, W. (2009). The impact of Levene's Test of equality of variances on statistical theory and practice. *Statistical Science*, 24(3), 343-360.

Gehring, T.M. & Swihart, R.K. (2002). Body size, niche breadth, and ecologically scaled responses to habitat fragmentation: Mammalian predators in an agricultural landscape. *Biological Conservation,* 109, 283-295.

Goldblatt, P. & Manning, J. 2002. *Cape Plants. A conspectus of the Cape Flora of South Africa.* National Botanical Institute, Pretoria.

Greef, G.J. (1991). *The geohydrology of a typical catchment in the Cape Supergroup, Breë River Valley.* Doctoral dissertation, Stellenbosch University, South Africa.

Hayward, M.W., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G. & Kerley, G.I.H. (2006). Prey preferences of the leopard (*Panthera pardus*). *Journal of Zoology*, 270, 298-313.

Hennig, R. (1998). *Schwarzwild: Biologie, verhalten, hege und jagd.* BLV Verlagsgesellschaft, Munich: BLV Publishing Company.

Hewitt, N. & Miyanishi, K. (1997). The role of mammals in maintaining plant species richness in a floating *Typha* marsh in southern Ontario. *Biodiversity and Conservation*, 6, 1085-1102.

Hoffman, T.S. & O'Riain, M.J. (2012). Troop size and human-modified habitat affect the ranging patterns of a chacma baboon population in the Cape Peninsula, South Africa. *American Journal of Primatology*, 74, 853-863.

Ishwaran, N., Persic, A. & Tri, N.H. (2008). Concept and practice: A case of UNESCO Biosphere Reserves. *International Journal of Environment and Sustainable Development*, 7(2), 118-131.

Jones, J.P.G., Andriamarovololona, M.M., Hockley, N., Gibbons, J.M. & Milner-Gulland, E.J. (2008). Testing the use of interviews as a tool for monitoring trends in the harvesting of wild species. *Journal of Applied Ecology*, 45, 1205-1212.

Kamler, J.F. & Macdonald, D.W. (2006). Longevity of a wild bat-eared fox. *South African Journal of Wildlife Research*, 36(2), 199-200.

Kamler, J.F., Stenkewitz, U., Klare, U., Jacobsen, N.F. & Macdonald, D.W. (2012). Resources partitioning among Cape foxes, bat-eared foxes, and black-backed jackals in South Africa. *The Journal of Wildlife Management,* 76(60), 1241-1253.

Kerley, G.I.H., Pressey, R.L., Cowling, R.M., Boshoff, A.F. & Sims-Castley, R. (2003). Options for the conservation of large and medium-sized mammals in the Cape Floristic Region hotspot, South Africa. *Biological Conservation*, 112, 169-190.

Keselman, H.J., Othman, A.R., Wilcox, R.R. & Fradette, K. (2004). The new and improved two-sample t test. *Psychology Science*, 15, 47-51.

Karanth, K. U. & Chellum, R. (2009). Carnivore conservation at the crossroads. F*auna & Flora International, Oryx,* 43(1), 1-2.

Khan, U., Lovari, S., Ali Shah, S. & Ferretti, S. (2018). Predator, prey and humans in a mountainous area: Loss of biological diversity leads to trouble. *Biodiversity and Conservation*, 27(11), 2795-2813.

Klein, R.G. & Cruz-Uribe, K. (1996). Size variation in the rock hyrax (*Procavia capensis*) and late quaternary climatic change in South Africa. *Quaternary Research*, 46, 193-207.

Lambeck, R.J. (1997). Focal species: A multi-species umbrella for nature conservation. *Conservation Biology*, 11(4), 849-856.

Leeney, R.H. (2015). Fishers' ecological knowledge of sawfish in Lake Piso, Liberia. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 26: 381–385.

Lessa, I., Guimaraes, T.C.S., Bergallo, H.G., Cuha, A. & Vieira, E.M. (2016). Domestic dogs in protected areas: A threat to Brazilian mammals? *Brazilian Journal of Nature Conservation*, 14, 46-56.

Lewis, J.S., Farnsworth, M.L., Burdett, C.L., Theobald, D.M., Grey, M. & Miller, R.S. (2017). Biotic and abiotic factors predicting the global distribution and population density of an invasive large mammal. *Scientific Reports*, 7.

Liberg, O. (1984). Food habits and prey impact by feral and house-based domestic cats in a rural area in Southern Sweden. *Journal of Mammalogy*, 65(3), 424-432.

Lindsey, P.A., Balme, G., Becker, M., Begg, C., Bento, C., Bocchino, C., et al. (2013). The bushmeat trade in African savannas: Impacts, drivers, and possible solutions. *Biological Conservation*, 160, 80-96.

Lloyd, P.H. (2000). Mammals. In: Turner, A.A. (Ed.), *Western Cape Province State of Biodiversity 2017* (pp. 1-19). Stellenbosch: CapeNature Scientific Services.

Lombard, A.T., Cowling, R.M., Pressey, R.L. & Mustart, P.J. (1997). Reserve selection in a species-rich and fragmented landscape on the Agulhas Plain, South Africa. *Conservation Biology*, 11(5), 1101-1116.

Lumney, M., Jones, A., Stiles, E. & Waltner-Toews, D. (2011). Preventative Veterinary Medicine, 102, 315-320.

Mann, G.K.H., Wilkinson, A., Hayward, J., Drouilly, M., O'Riain, M.J. & Parker, D.M. (2019). The effects of aridity on land use, biodiversity and dietary breadth in leopards. *Mammalian Biology*, 98, 43-51.

Marker, L.L. & Dickman, A.J. (2005). Factors affecting leopard (*Panthera pardus*) spatial ecology, with reference to Namibian farmlands. *South African Journal of Wildlife Research*, 35(2), 105-115.

Martins, Q., Horsnell, W.G.C., Titus, W., Rautenbach, T. & Harris, S. (2010). Diet determination of the Cape Mountains leopards using global positioning system location clusters and scat analysis. *Journal of Zoology*, 283, 81-87.

Martins, Q. & Martins, N. (2006). Leopards of the Cape: Conservation and conservation concerns. *International Journal of Environmental Studies*, 63(5), 579-585.

Massara, R.L. & Chiarello, A.G. (2012). Is the domestic dog becoming an abundant species in the Atlantic Forest? A case study in southeastern Brazil. *Mammalia*, 76, 67-76.

Matthiae, P.E. & Stearns, F. (1981). Mammals in forest islands in southeastern Wisconsin. In R.F. Whitcomb, R.L., Burgess & D.M. Sharpe (Eds.), *Forest Island Dynamics in Man-dominated Landscapes* (pp. 55-66). New York: Springer-Verlag.

McLaughlin, A. & Mineau, P. (1995). The impact of agricultural practices on biodiversity. *Agriculture, Ecosystems and Environment*, 55, 201-212.

Milton, S.J. & Dean, W.R.J. (2000). Seed dispersal in dung of insectivores and herbivores in semi-arid southern Africa. *Journal of Arid Environments*, 47, 465-483.

Morrison, J.C., Sechrest, W., Dinerstein, E., Wilcove, D.S. & Lamoreux, J.F. (2007). Persistence of large mammal faunas as indicators of global human impacts. *Journal of Mammalogy*, 88(6), 1363-1380.

Nelner, T.B. & Hood, G.A. (2011). Effect of agriculture and presence of American beaver Castor Canadensis on winter biodiversity of mammals. *Wildlife Biology*, 17, 326-336.

Newmark, W.D. (2008). Isolation of African protected areas. *Frontiers in Ecology and the Environment*, 6(6), 321-328.

Nieman, W.A., Leslie, A.J., Wilkinson, A. & Wossler, T.C. (2019). Socioeconomic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa. *Journal of Nature Conservation*, 52, 1-7.

Norton, P.M., Lawson, A.B., Henley, S.R. & Avery, G. (1986). Prey of leopards in four mountainous areas of the south-western Cape Province. *South African Journal of Science and Technology*, 16(2), 47-52.

Norton, P.M. & Lawson, A.B. (1985). Radio-tracking of leopards and caracals in the Stellenbosch area, Cape Province. *South African Journal of Wildlife Research*, 15(1), 17-24.

Noss, A.J. (1998). The impacts of cable snare hunting on wildlife populations in the forests of the Central African Republic. *Conservation Biology*, 12, 390–398.

Nyhus, P.J., Tilson, S. & Tilson, R. (2003). Wildlife knowledge among migrants in southern Sumatra, Indonesia: Implications for conservation. *Environmental Conservation*, 30(2), 192-199.

Oberosler, V., Groff, C., Lemma, A., Pedrini, P. & Rovero, F. (2017). The influence of human disturbance on occupancy and activity patterns of mammals in the Italian Alps from systematic camera trapping. *Mammalian Biology*, 87, 50-61.

O'Connell Jr., A.F., Talancy, N.W., Bailey, L.L., Sauer, J.R., Cook, R. & Gilbert, A.T. (2006). Estimating site occupancy and detection probability parameters for meso- and large mammals in a coastal ecosystem. *The Journal of Wildlife Management*, 70(80), 1625-1633.

Ott, T., Kerley, G.I.H & Boshoff, A.F. (2007). Preliminary observations of the diet of leopards (*Panthera pardus*) from a conservation area and adjacent rangelands in the Baviaanskloof region, South Africa. Journal of African *Zoology*, 42(1), 31-37.

Owen-Smith, N. & Mills, M.G.L. (2007). Predator–prey size relationships in an African large-mammal food web. *Journal of Animal Ecology*, 77(1), 173-183.

Palmer, R. & Fairall, N. (1988). Caracal and African wildcat diet in the Karoo National Park and the implications thereof for hyrax. *South African Journal of Natural Science*, 18(1), 30-34.

Pardini, R., Nichols, E. & Püttker, T. (2018). Biodiversity response to habitat loss and fragmentation. In D.A. Dellasala & M.I. Goldstein (Eds.), *Encyclopedia of the Anthropocene* (pp. 229-239). Oxford: Elsevier.

Park, K.J. (2014). Mitigating the impacts of agriculture on biodiversity: Bats and their potential role as bioindicators. *Mammalian Biology*, 80, 191-204.

Pearce, J.L. & Boyce, M.S. (2006). Modelling distribution and abundance with presence-only data. *Journal of Applied Ecology*, 43, 405-412.

Pettorelli, N., Labora, A.L., Msuha, M.J., Foley, C. & Durant, S.M. (2010). Carnivore biodiversity in Tanzania: Revealing the distribution patterns of secretive mammals using camera traps. *Animal Conservation*, 13, 131-139.

Picker, M. & Griffiths, C. (2011). *Alien and invasive animals: A South African Perspective*. Cape Town: Struik Nature.

Pillay, R., Johnsingh, A.J.T., Raghunath, R. & Madhusudan, M.D. (2011). Patterns of spatiotemporal change in large mammal distribution and abundance in the southern Western Ghats, India. *Biological Conservation,* 144, 1567-1576.

Pool-Stanvliet, R. (2013). A history of UNESCO man and the biosphere program in South Africa. *South African Journal of Science*, 109 (9-10):01-06.

Pool-Stanvliet, R. & Giliomee, J.H. (2013). A sustainable development model for the winelands of the Western Cape: A case study of the Cape Winelands Biosphere Reserve. *AfriMAB*, 45-71.

Pool-Stanvliet, R., Duffell-Canham, A., Pence, G. & Smart, R. (2017). The Western Cape spatial plan handbook. Stellenbosch: Cape Nature.

Quinn, R.D. (1986). Mammalian herbivory and resilience in Mediterranean-climate ecosystems. Dell B., Hopkins A.J.M. & Lamont B.B. (Eds.), *Resilience in Mediterranean-type ecosystems (pp)*. Tasks for Vegetation Science, 16.

Radloff, F.G.T., Mucina, L., Bond, W.J. & le Rous, P.J. (2010). Strontium isotope analyses of large herbivore habitat use in the Cape Fynbos region of South Africa. *Oecologia,* 164, 567-578.

Rautenbach, T. (2010). *Assessing the diet of Cape leopard (Panthera paradus) in the Cederberg and Gamka mountains, South Africa*. Unpublished master's thesis, Nelson Mandela Metropolitan University.

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L., Rutherford, M.C., Smit, W.J. et al. (2006). Fynbos Biome. In: L. Mucina & M.C. Rutherford (Eds.), *The Vegetation of South Africa, Lesotho and Swaziland* (pp. 52–219). Pretoria: SANBI.

Rogan, J.E. & Lacher Jr., T.E. (2018). Impacts of habitat loss and fragmentation on terrestrial biodiversity. *In Reference Module in Earth Systems and Environmental Sciences*. Amsterdam: Elsevier.

Rouget, M., Richardson, D.M., Cowling, R.M., Lloyd, J.W. & Lombard, A.T. (2003). Current patterns of habitat transformation and future threats to biodiversity in terrestrial ecosystems of the Cape Floristic Region, South Africa. *Biological Conservation*, 112, 63-85.

Rutherford, M.C., Mucina, L. & Powrie, L.W. (2002). Nama-karoo veld types revisited: A numerical analysis of original Acocks' field data. *South African Journal of Botany,* 69(1), 52-61.

Schoener, T.W. (1982). The controversy over interspecific competition. *American Scientist*, 70, 586-595.

Schuman, H. & Scott, J. (1987). Problems in the use of survey questions to measure public opinion. *Science, New Series*, 236(4840), 957-959.

Serieys, L.E.K., Bishop, J., Okes, N., Broadfield, J., Winterton, D.J., Poppenga, R.H. et al. (2019). Widespread anticoagulant poison exposure in predators in a rapidly growing South African city. *Science of the Total Environment*, 666, 581-590.

Shingala, M.C. & Rajyaguru, A. (2015). Comparison of post hoc tests for unequal variance. *International Journal of New Technologies in Science and Engineering,* 2(5), 22-33.

Shiponeni, N.N. & Milton, S.J. (2005) Seed dispersal in the dung of large herbivores: Implications for restoration of Renosterveld shrubland old fields. *Biodiversity and Conservation*, 15, 3161-3175.

Silva-Rodríguez, E.A. & Sieving, K.E. (2012). Domestic dogs shape the landscape-scale distribution of a threatened forest ungulate. *Biological Conservation*, 150, 103-110.

Sinclair, A.R.E. (2003). The role of mammals as ecosystem landscapers. *Alces*, 39, 161-176.

Skinner, J.D. & Chimimba, C.T. (2005). The mammals of the southern African subregion. 3rd edition. Cape Town: Cambridge University Press.

Spatial Planning (2018). Western Cape land-use planning: Rural guidelines. Draft report for public comment: January 2018, Cape Town.

Statistics South Africa (2018). *Mid-year population estimates 2018, Statistical Release P0302*, Statistics South Africa, Pretoria.

Stein, A., Athreya, V., Gerngross, P., Balme G., Henschel, P., Karanth U. et al. (2016). Leopard *Panthera pardus* [Online]. Retrieved August 24, 2018: https://www.iucnredlist.org/species/15954/102421779

Stenkewitz, U., Herrmann, E. & Kamler, J.F. (2010). Distance sampling for estimating springhare, Cape hare and steenbok densities in South Africa. *South African Journal of Wildlife Research*, 40(1), 87-92.

Stevens, U. (2003). *The Winelands explorer*. Cape Town: Wanderlust Books.

Stuart, C.T., Stuart, T. & Pereboom, V. (2003). Diet of bat-eared fox (Otocyon megalotis), based on scat analysis, on the Western Escarpment, South Africa. *Canid News*, 6 (2).

Stuart, C. & Stuart, M. (2015). Stuart's field guide to mammals of Southern Africa. Struik Nature, Cape Town.

Swanepoel, L.H., Lindsey, P., Somers, M.J., van Hoven, W. & Dalerum, F. (2013). Extent and fragmentation of suitable leopard habitat in South Africa. *Animal Conservation*, 16, 41-50.

Taylor, W.A. (2005). *Factors influencing productivity in sympatric populations of Mountain Reedbuck and Grey Rhebok in the Sterkfontein Dam Nature Reserve, South Africa*. Unpublished doctoral dissertation, University of Pretoria.

Taylor, A., Cowell, C. & Drouilly, M. (2017). Pelea capreolus. The IUCN Red List of Threatened Species 2017, Retrieved from http://dx.doi.org/10.2305/IUCN.UK.2017-2.RLTS.T16484A50192715.en

Tizora, P., Le Roux, A., Mans, G. & Cooper, A. (2016). *Land use and land cover change in the Western Cape Province: Quantification of changes & understanding of driving factors.* Unpublished paper delivered at the Seventh Planning Africa Conference on Making Sense of the Future: Disruption and Reinvention. Johannesburg, July 4-6.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I. & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters*, 8, 857-874.

Vanak, A.T. & Gompper, M.E. (2010). Interference competition at the landscape level: The effect of free-ranging dogs on a native mesocarnivore. *Journal of Applied Ecology*, 47, 1225–1232.

Van Wilgen, B.W. (1987). Fire regimes in the Fynbos biome. In R.M. Cowling, D.C. Le Maitre, B. McKenzie, R.P. Prys-Jones & B.W. van Wilgen (Eds.), *Disturbance and the dynamics of Fynbos Biome communities* (pp. 6-12). Pretoria: Foundation for Research Development.

Verberk, W.C.E.P. (2011). Explaining general patterns in species abundance and distributions. *Nature Education Knowledge*, 3(10), 38.

Von Hase, A., Rouget, M. & Cowling R.M. (2010). Evaluating Private Land Conservation in the Cape Lowlands, South Africa. *Conservation Practice and Policy*, 24(5), 1182-1189.

White, P.C.L., Jennings, N.V., Renwick, A.R. & Barker, N.H.L. (2005). Questionnaires in ecology: A review of past use and recommendations for best practice. *Journal of Applied Ecology*, 42, 421-430.

Wimberger, K., Downs, C.T. & Perrin, M.R. (2009). Two unsuccessful reintroduction attempts of rock hyraxes (*Procavia capensis*) into a reserve in the KwaZulu-Natal Province, South Africa. *South African Journal of Wildlife research,* 39(2), 192-201.

Woodroffe, R., & Donnelly, C.A. (2011). Risk of contact between endangered African wild dogs *Lycaon pictus* and domestic dogs: Opportunities for pathogen transmission. *Journal of Applied Ecolo*gy, 48, 1345-1354.

Young, J.K., Olsen, K.A., Reading, R.P., Amgalanbaatar, S. & Berger J. (2011). Is wildlife going to the dogs? Impacts of feral and free-roaming dogs on wildlife. *Bioscience*, 61(2), 125-132.

Appendix 3.1: The questionnaire used when interviewing the owners and managers of each farm

Medium-sized mammals survey owner/managers version

(INTERVIEWERS USE)

Location:…………………………………………………… Time:…………………………

Interviewee identification:……………………………………………………………………

Study site and background (Emailed before hand, self fill in)					
1.	Property name				
2.	GPS location				
3.	Deed number				
4.	Property size (ha)				
5.	Percentage area land-use (%)	Livestock:	Orchard:	Vineyard:	Conservation:
		Natural vegetation:	Forestry:	Other:	What?
6.	Conservancy name:				
7.	Owner of the farm/tenure:				

SECTION A: Socio-economic information						
1	Population group	Asian	Black	Coloured	White	Other
2	Gender	Female		Male		
3	Age					
4	Home Language	Afrikaans	English	Xhosa	Other	
5	Job title					
6	No. of years present on the property?					
7	Where did you grow up?					
8	Where do you live?	On the property	A neighbouring property	in a town		

0

Appendix 3.2: Mammal survey questions asked for both managers and labourers

SECTION C: Medium-sized Mammal Abundance

1.) Complete Table 1 about which wild mammals in the images provided, you have seen on this property and how their populations have changed since you've been present on the property?

					When?					Population changes						
Animal Type	Presence	Max. no.	Min. no.	Freq.	1 year	1-5 years	5-10 years	10-20 years	+20 years	Increase	Decrease	Fluctuate	Stable	Uncertain	N/A	Why?
Leopard																
Caracal																
African wildcat																
Feral domestic cat																
Cape fox																
Bat-eared fox																
Chacma baboon																
Aardwolf																
Aardvark																
Honey badger																
Porcupine																
Rock hyrax																
Common duiker																
Grey rhebok																
Grysbok																
Klipspringer																
Bushbuck																
Uncertain of buck																

1

					When?					Population changes						
Animal Type	Present	Max. no.	Min. no.	Freq.	1 year	1-5 years	5-10 years	10-20 years	+20 years	Increase	Decrease	Fluctuate	Stable	Uncertain	N/A	Why?
Cape clawless otter																
Water mongoose																
Large grey mongoose																
Cape grey mongoose																
African striped weasel																
Striped Polecat																
Genet																
Bush pig																
Feral pig																
Hare																
Red rock rabbit																
Feral dog																

Appendix 3.3: The questionnaires used when interviewing the labourers of each farm

Medium-sized mammals survey Labourers version

Location:.. Time:...

Interviewee identification:...

	SECTION A: Socio-economic information					
1	Population group	Asian	Black	Coloured	White	Other
2	Gender	Female		Male		
3	Age					
4	Home Language	Afrikaans	English	Xhosa	Other	
5	Job title					
6	Family size					
7	No. of years present on the property?					
8	Where did you grow up?					
9	Where do you live?	On property	Neighbouring property	Town	Informal settlement	
		Other: (State what)				
	SECTION B: Medium-sized mammal Abundance					
1	What animals have you seen on the property?					
2	Complete Table 1 about which wild mammals in the images provided, you have seen on this property and how their populations have changed since you've been present on the property?					

1

Appendix 3.4: Letter of consent to be interviewed that was given to each study participant prior to their approved interview. Consent letters were also available in Afrikaans and isiXhosa.

Letter of consent to be interviewed

Date:
ID:

Dear Participant, this is a letter of consent stating that you agree to participate in this interview as part of a study investigating presences, abundances and human-related threats to medium-sized mammals in the Cape Winelands and Overstrand areas.

Your contribution to the study is important and knowledge which you have on these topics is unattainable anywhere else. This study will assess what current distributions and abundances of mammals and whether any population changes have occurred and whether threats do exist. This will aid us in how to deal with threats so that all stakeholders including you, can better benefit from the wildlife in this area.

The interview will take place in a mutually agreed upon location on the property. You shall remain anonymous in the study; no name will be required, and all information is considered completely confidential. Your participation is voluntary. If at any point you want to stop the interview, you may withdraw without any consequences. We only ask that you are honest with all your answers and if you are uncomfortable with answering any of the questions that you'd simply decline to answer.

We ask your permission to use the information shared by you anonymously for our study analysis and possibly in future publications. Please feel free to give as much information as you have and ask any questions of your own. There are no known or anticipated risks to you as a participant in this study.

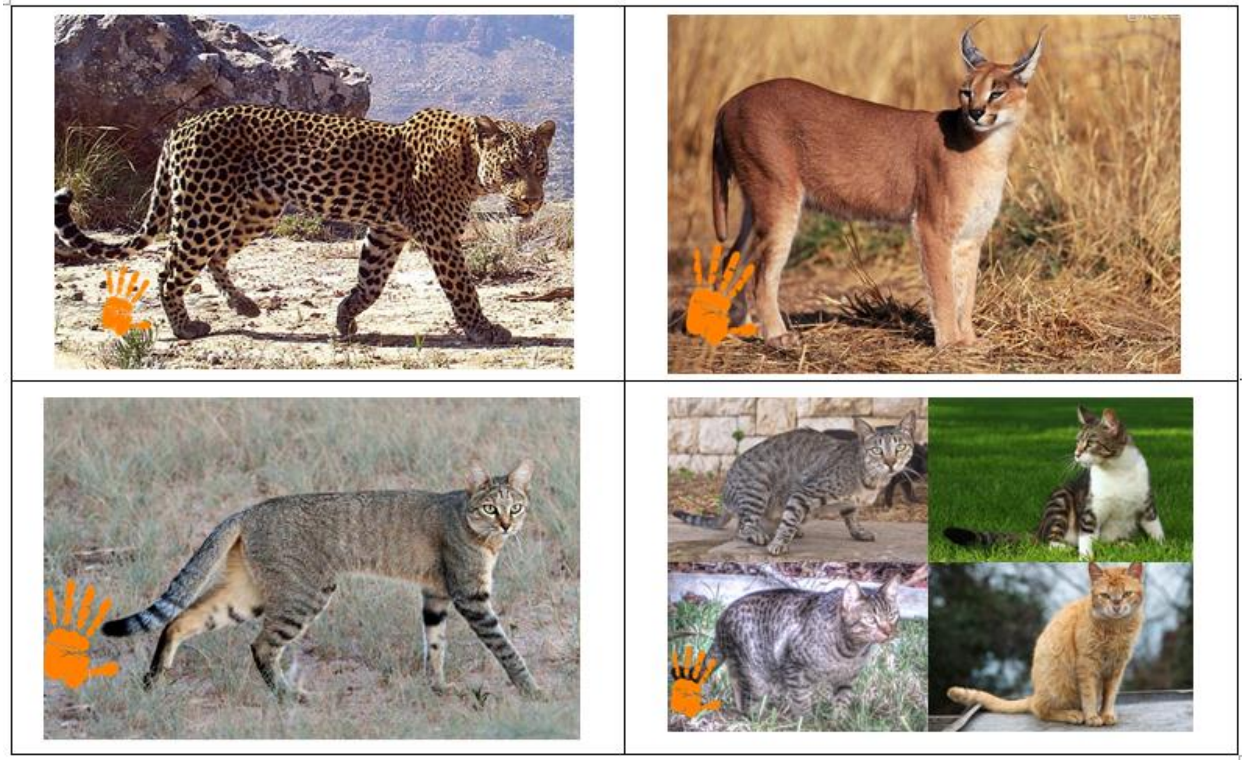
☐ Gives consent to conduct study
☐ Does NOT give consent to conduct study

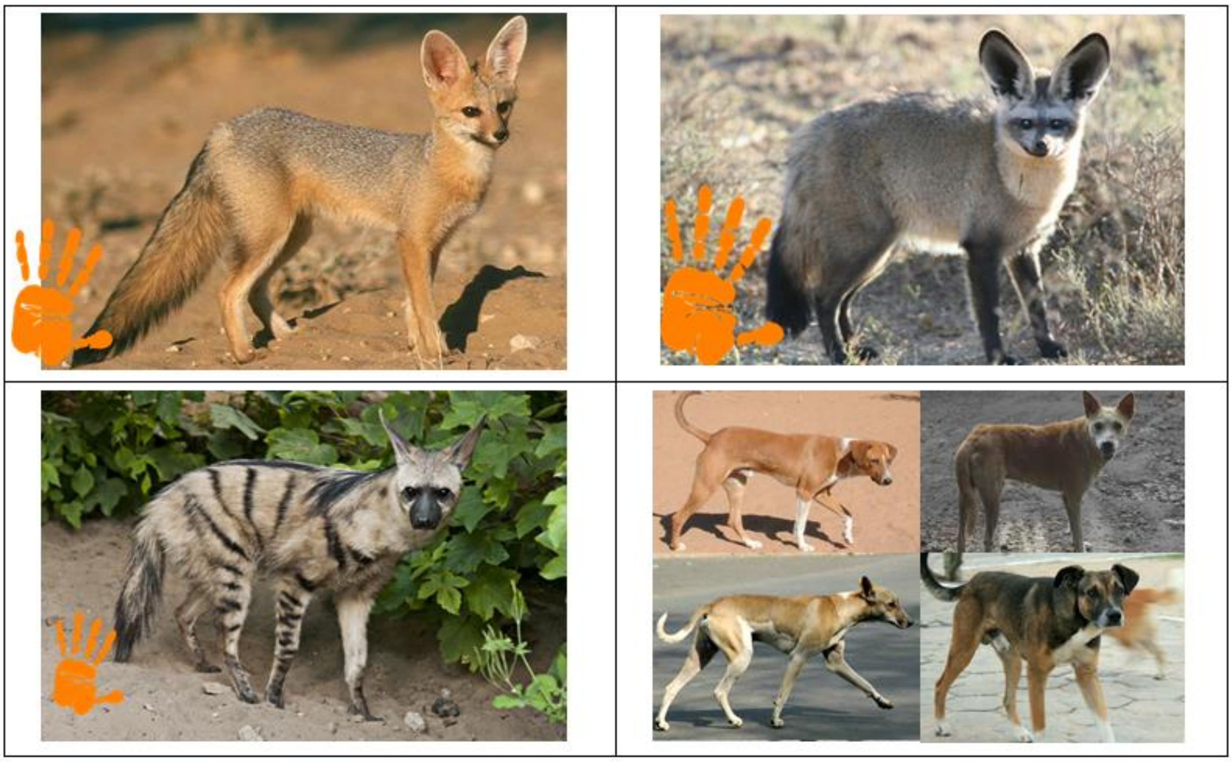
If you have any further questions or want any further information regarding this, please contact:
Anita Wilkinson (The Cape Leopard Trust): 071 [redacted] / [redacted]
Brittany Schultz (MSc candidate – Stellenbosch University): 079 [redacted] / 17838908@sun.ac.za

Appendix 3.5: List of study species, their scientific and other names, and images used on cue cards

Name	Scientific name	Other names
Leopard	*Panthera pardus pardus*	Luiperd , Tier,
Caracal	*Caracal caracal*	Rooikat, lynx,
African wild cat	*Felis silvestris lybica*	Vaalboskat
Feral domestic cat	*Felis catus*	Ronloper kat, stray cat
Cape fox	*Vulpus chama*	Silwervos, draai jakkals
Bat-eared fox	*Otocyon megalotis*	Bakoorjakkals, bakoorvos
Aardwolf	*Proteles cristata*	Aardwolf
Feral domestic dog	*Canis lupus familiaris*	Rondloperhond, stray dog
Honey badger	*Mellivora capensis*	Ratel,
Water mongoose	*Atilax paludinosus*	Kommetjiegatmuishond, marsh mongoose
Large grey mongoose	*Herpestes ichneumon*	Groot grysmuishond, Egyptian mongoose, Ichneumon
Cape grey mongoose	*Galerella pulverulenta*	Klein grysmuishond, small grey mongoose
Large OR small spotted genet	*Genetta tigrine* / *Genetta genetta*	Muskejaakat, genet
Cape clawless otter	*Aonyx capensis*	Groototter
African striped weasel	*Poecilogale albinucha*	Slangmuishond, White-naped weasel
Striped polecat	*Ictonyx striatus*	Stinkmuishond, African polecat, zorilla, skunk,
Chacma baboon	*Papio ursinus*	Kaapse bobejaan
Aardvark	*Orycteropus afer*	Aardvark, antbear
Bush pig	*Potamochoerus larvatus*	Bosvark
Feral pig	*Sus scrofa*	Wilde vark, wild boar
Cape grysbok	*Raphicerus melanotis*	Grysbok
Klipspringer	*Oreotragus oreotragus*	Klipspringer
Common duiker	*Sylvicapra grimmia*	Gewone duiker, grey duiker
Grey rhebok	*Pelea capreolus*	Vaalribbok, Vaal Rhebok
Rock hyrax	*Procavia capensis*	Klipdas, rock dassie
Bushbuck	*Tragelaphus sylvaticus*	Bosbok,
Cape porcupine	*Hystrix africaeaustralis*	Ystervark, African porpcupine
Cape OR scrub hare	*Lepus capensis* / *Lepus saxatilis*	Haas, vlakhaas, kolhaas, hare
Smith's red rock rabbit	*Pronolagus rupestris*	Smith se rooiklipkonyn

Appendix 3.6: Cue cards presented to interviewees of each study species











Medium-Sized Mammals of the Study Area

The following cards contain mammals you may have seen in the area.
The hand symbol acts as a common scale and represents a 20cm long hand
Common Afrikaans, English and other names are on the back of each card

Chapter 4

General discussion, conclusions and future management recommendations

4.1 Overview

This study presents an initial approach to two very complex, large and fundamental factors contributing to the survival of leopard and medium-sized mammals in the Boland Mountain Complex (BMC), Western Cape Province. The primary goal of this study was to assess whether and/or how human development threatens leopard, their medium-sized mammalian prey and both their habitat availability in agricultural buffer zones of the BMC. The study aimed to assess whether medium-sized mammal population distribution and abundance, fire regimes and land-covers had changed over the most recent 30-year time-period and further to assess whether identified changes require priority in terms of further research and/or management resources.

The objective of Chapter Two was to determine whether habitat available to medium-sized mammals, had changed over 23-years due to land-cover and/or fire regime shifts over time, and if so where. From 1990 to 2013, slightly more vegetation was available to act as medium-sized mammalian habitat and few changes to land-covers that could negatively impact mammals were detected on the scale of the BMC. Shifts in fire regimes, from 1957 to 2017 displayed increased frequencies and sizes of areas of total land burning per year, confirming findings of similar studies (e.g. Van Wilgen et al., 2010). It is therefore, probable that fire regime shifts over time may have lowered the functionality of many habitats utilised by mammals during fire disturbances.

Chapter Three aimed to establish whether sightings of any medium-sized mammal populations had changed over time and if these differed across the agricultural buffer zones. Differences in abundances of species were observed throughout the agricultural buffer zones. These agro-ecosystems are utilised by all the mentioned study species, which exposes them to an array of anthropogenic threats, and therefore location specific management is needed. Leopards' highly adaptive behaviour may allow for their persistence through the current habitat and prey availability, but their use of habitat within agro-ecosystems exposes them to a synergy of mortality risks.

Key results in this study and management implications are further discussed in detail below.

4.2 Key findings

4.2.1 Land-cover changes

From 1990 to 2013, 107.6km^2 more total habitat, represented by vegetation, became available in the study area. This was due to a large area of 146.76km^2 of plantations that were felled and cleared. No other land-covers showed significant shifts and agricultural expansion did not exhibit a major threat to medium-sized mammals during this period. Land-cover changes to vegetation appeared as several small patches, within agriculture matrices and only a few large expanses adjacent to the core. Some differences were visible between the seven Mountain Zones (MZ). Simonsberg MZ was an outlier, with the smallest proportion of vegetation and largest percentage of agriculture. This MZ had the most vegetation converted to agriculture from 1990 to 2013. The East Hawequas MZ was the only mountain area to have agricultural land-cover increase overall. The West Hottentots-Holland MZ had the greatest increase in built-up land-cover. Both biosphere reserves showed similar land-cover compositions and showed limited differences in the land-cover changes over time. Agriculture did replace proportionally more vegetation in the Cape Winelands Biosphere Reserve (CWBR) than in the Kogelberg Biosphere Reserve (KBR). It was built-up areas that replaced more habitat in the KBR.

4.2.2 Fire regime changes

The fire regime had been altered substantially by the 1990's. Approximately three times more area was burnt from 1987 to 2017 than from 1957 to 1987. The rate at which the majority of available habitat was burning meant that for the 15-year period from 2002 to 2017, it would take as little as 20 years for the entire area to burn at least once. The fire data showed that for the 1957 to 1972 period it was taking up to 120 years for the entire study area to burn. From 1987 to 2017, an average of 149.92km^2 of total land-cover (mostly vegetation and forestry) was burnt per year. Whereas in the previous 30 years (from 1957 to 1987) an average of only approximately 48.77km^2 of land-cover area was burnt per year. The number of fire ignitions per fire year have drastically increased and the majority of ignitions are from human sources.

In the earliest periods, from 1957 to 1972, there was great variance in the Fire Return Intervals (FRI) between each MZ. All MZs had FRI's near equal in length by 2002, with little variance across the study area. The majority of the BMC had been burnt at least twice in the 30-year period from 1987 to 2017, with some patches exceeding 10 burns during that period. In earlier periods each biosphere reserve had visibly great variances between the fire regimes. In later, most recent years, a similar fire regime was seen across both biosphere reserves. Those sections of the study area that were further from human settlements, had more larger areas burnt in single fires since 1957.

4.2.3 Medium-sized mammal perceived abundances

Medium-sized mammal compositions were perceived to remain relatively consistent over time in the BMC. Baboon populations on farms closer to settlements were more often perceived as decreasing and rock hyrax populations showed significant decreases closer to major roads. Caracal populations were perceived as stable throughout the agro-ecosystems.

Between the two biosphere reserves, grey rhebok, hares and feral pig showed variations in perceived abundances. Feral pigs were present on more farms, while rhebok and hares were present on fewer farms in the CWBR than in the KBR. Many species varied in perceived abundance among the seven defined MZs. Simonsberg and the West Hottentots-Holland MZs had fewer farms with baboon, hares, klipspringer, rhebok and large grey mongoose present. Groenlandberg MZ's perceived abundances of rock hyrax was lower than those in any other MZ. The East Hawequas MZ had a higher perceived abundance for many of the study species than anywhere else.

Recorded species' presence was influenced by a property's distances from the various land-covers. Leopard, duiker, klipspringer, hares and large grey mongoose showed widespread presence on farms further from settlement edges. Other species occurred further from major roads, including rhebok, hares, feral pig, African wildcat, feral cat, bat-eared fox, genet and honey badger. Smith's red rock rabbit however, was seen on farms closer to roads. The distance from official Protected Area (PA) boundaries affected the distribution of klipspringer, genet and polecat that were seen on farms closer to these edges. Rock hyrax, hares, baboon, feral dogs and African striped weasels were present on farms at greater distances from PA boundaries.

The total percentage of natural vegetation on farms was correlated to the presence of a few species. Leopard, klipspringer and rock hyrax were present on more farms with a greater percentage of natural vegetation cover. On the other hand, duikers were more present on farms displaying a lower percentage of natural land-cover.

Feral cats were detected on many farms as well as feral dogs and feral pigs. Feral pigs were a detected pest in Franschhoek and further north in Wellington, as previously reported (Birss, 2017; Hignett, 2006), but were again detected on farms further south-east, near Botriver and Villiersdorp. Feral dogs were present at similar perceived abundances across the agro-ecosystems.

4.2.4 Detected concerns

Habitat loss in terms of quality and accessibility is a more likely concern for medium-sized mammals and leopard than the physical quantitative change in land-covers displayed from 1990 to 2013 in the BMC. Functional corridors within the BMC and into other mountain ranges are at risk of being reduce in size or being completely closed off with continued habitat loss. The isolation of PAs from large core areas is concerning in terms of the persistence of mammals within them. Key or rare medium-sized mammalian habitats (such as lowland fynbos, renosterveld, grassland and sand fynbos) are also threatened within agricultural buffers (Radloff, 2008).

The size, quality and accessibility of habitat refuges, utilised by medium-sized mammals, are threatened by the more frequent fires and larger total areas burning per year in recent periods

(Kelly et al., 2012). These fire regimes may cause insufficient regeneration of riparian vegetation or lead to too few unburnt similar refuges per year (Driscoll et al., 2010). It is possible that the continuation of the fire regime patterns that this study has detected may alter the natural vegetation composition and its ability to support medium-sized mammals (Kraaij & Novellie, 2010). The homogenous burn patterns displayed may also lower the diversity of habitat types available to mammals (Ricketts & Sandercock, 2016). The number of ignitions were mostly and increasingly from human-caused sources.

Populations of naturally occurring mammals for which the highest concern is reported here, for their future persistence in the BMC, were grey rhebok and the Cape and scrub hares. The CWBR showed more negative results than the KBR. The more negative results for the CWBR can be attributed to two of the MZs. Simonsberg and the West Hottentots-Holland MZs showed lower perceived abundances of multiple mammal species, more than the five other MZs. This could potentially be explained by the various negative impacts emitted by the larger human populations and greater sizes of human development in the towns of Stellenbosch, Franschhoek and Somerset West that border these MZs.

Built-up, forestry and agriculture land-cover classes produce various effects on medium-sized mammals and their habitats. It was the negative impacts that stem from these three land-cover classes which raised the greatest concerns for the BMC. Feral dogs are likely symptoms of exposure to human development that were detected throughout the agro-ecosystem landscape. Other reported impacts of the above-mentioned human land-covers in this study included: fences, roads, poaching, pest species removal and invasive vegetation. When moving through the BMC's agro-ecosystems to patrol their territories, hunt medium-sized mammalian prey or disperse, leopard may be exposed to all these above-mentioned threats and more, which may have a synergy of negative effects on their populations.

4.3 Conclusions

The investigation into land-cover change within a relatively short time period, presented a positive outcome. Namely, that more potential habitat has been made available from various anthropogenic land-covers, since 1990. When analysing the local fire regime, as an element of habitat loss, over a much longer period, a concern for habitats' continued ability to sustain medium-sized mammals was highlighted. The investigation of relative perceived medium-sized mammal abundances was one of only a few studies for most of the mammals, specifically within the BMC.

The average sized area that was burnt by 2013, has more than tripled since prior to the 1990's. The majority of the study area now burns within a 20-year period. However, from the 1950's to the 1980's, fires were more sporadic, and some areas may have been excluded from fire for over 100 years. Although, the increase in 107.6km^2 of vegetation land-cover is not a solution to the increased burnt area, it does provide additional habitat to accommodate mammals to use as corridors to access other habitats or refuge (Caudill et al., 2014).

Chapter two revealed that built-up land-covers were not greatly expanding into and threatening mammals' available habitat from 1990 and 2013. However, many direct and indirect effects emitted from human development and built-up land-covers need consideration. Results from Chapter Three indicated that settlement distances from farms negatively influence the perceived abundance of leopard, duiker, klipspringer, hare and large grey mongoose. The assortment of mammal species from various functional groups indicates these are relevant patterns to the species, and not an effect of bias from interviewees being more present near settlements. The exact factors that cause these higher perceived species' abundances closer to settlements is unclear. Major tarred roads are one of the prominent factors within human settlements. The ecological impacts that roads create potentially warded off hare, grey rhebok, genet and feral pig, but not the other above-mentioned species (Barthelmess, 2014). It may be because shorter distances between settlements increases access for humans or increases the magnitude of effects to mammalian habitats. These patterns in perceived abundances may be due to factors where settlements are established that are generally further from high gradients or rocky outcrops that are the preferred habitats for some species (Norton et al., 1986). This is apparent for leopard, klipspringer and rock hyrax because of their preference to farms with more natural vegetation that are not ideal for agriculture (Norton et al., 1986).

This supports how the gain of natural vegetation in the 23-year period can benefit multiple species. As Caudill et al. (2014) suggests, it may aid in agriculture forming a sustainable buffer matrix. Greater expanses of natural vegetation provide better shelter and hold more resources for mammals in general (Driscoll et al., 2010). Therefore, any sized, continuous, natural habitat extending into buffer land-covers may benefit a mammal's range and it is important to maintain connectivity to more habitats (Caudill et al., 2014; Woodroffe & Ginsberg, 1998).

Although baboons, duiker and Cape grey mongoose likely benefit from agriculture, species that move further onto farms may encounter greater risks of human-wildlife conflict. Hare, rock hyrax, baboon and feral dogs are reported pests in some agricultural areas (Farfán et al., 2011; Fehlmann et al., 2017; Moran et al., 1987; Stuart & Stuart, 2013; Stuart & Stuart, 2015). The limited expansion of agriculture into natural habitat may not affect population survival, but movement onto accessible agricultural areas may expose the species to increased mortality risks (Tscharntke et al., 2005). Agricultural expansion may be limited and insignificant because of recent intensification of farming practices, but this practice could bring about other threats by creating a less accessible landscape with stronger barriers and intense effects (Matson et al., 1997; Sotherton, 1998).

The large areas of forest cleared are likely beneficial to natural biodiversity (Ruiz, 2003), however, without rehabilitation the felled plots would not support the same biodiversity (Louw, 2010). Clearing pine plantations should allow for a more natural fire regime (Kraaij et al., 2013), functional movement corridors (Golley et al., 1975) and higher diversity of habitat types (Rebelo et al., 2019).

The high diversity and high perceived abundances of some species in agricultural buffers alleviates some concern over low leopard prey availability. Threats to medium-sized mammals in buffer habitats seems like a negligible threat to highly adaptive leopard but may have other indirect effects on the ecology of the BMC (Henschel et al., 2011). Mesopredators with more specialised diets may be greatly affected by the lower perceived abundances of some species within sections of the landscape (Prugh et al., 2008). Based on chance observations, it was grey rhebok and hares that drove the most concern for their variations across the landscape.

Simonsberg stood out as the smallest MZ and its core was completely encircled by agriculture, main roads and human development. These may have contributed to Simonsberg MZ's fewer fires occurring in the area. The small, isolated size of the remaining natural area could reduce the diversity of habitat types, which may have reduced the capacity of mammal diversity (Crooks, 2002; Tscharntke et al., 2005). Prugh et al. (2008) described how isolation and small size of habitat patches affected animal diversity. This is potentially why multiple study species had lower perceived abundances in this MZ.

Although the changes in size of available habitat was of low concern to mammals in this study, the quality of that space needs further understanding. This study was unable to rule out whether the quality of habitat had remained, increased or decreased over time and whether components of its integrity were able to support all the mentioned study species. The few differences and inconsistencies between species may be from the quality of these habitats. The land-cover data could not differentiate between natural and alien vegetation stands, which do need specific consideration. The clearing of alien vegetation (including various *Acacia sp*. and *Hakea sp*.) was an activity mentioned by a few private properties as having changed what mammal compositions were seen. Most stated that clearing increased the number of mammal and bird species seen on the properties and some believed it was the reason as to why baboons spent more time in the new natural vegetation rather than in vineyards. The locations of functional and threatened habitats are important to consider. This study could however not detect the quality of movement corridors.

Having human developments so entwined with natural land-covers, allows for open access for people to utilise these habitats. Uses, like ecotourism, may have similar benefits to both the PA's and humans. Many uses are however unregulated, unsustainable, only benefit some individuals and may be the cause for declines seen in some mammal populations. The removal of pest animals due to human-wildlife conflicts, for entertainment or for uses as food, clothing, accessories or traditional medicine have been documented in many places in South Africa (Becker et al., 2012; Hayward, 2009). Many incidences of poaching with wire snares and packs of dogs were documented during data collection, affecting multiple study species (Nieman et al., 2019). Informal hunting with wire snares was undertaken by some labourers, their families and other people who could access the farms, supposedly intended to capture specific mammal species for various uses, but without any distinction between what is caught (Hayward, 2009; Nieman et al., 2019).

The repercussions of land-cover change from natural habitats to human developments introduce and expose mammals to further threats that intensify the impacts of physical loss to habitat (Newmark, 2008). Roads are narrow features that should fall under the built-up land-cover class. They replace habitat and alter its structure along their edges too (Shepard et al., 2008). They act as complete barriers for some species movement patterns, interrupting gene flow and fragmenting natural landscapes (Kioko et al., 2015; Rico et al., 2007). A direct effect is wildlife-vehicle collisions that can result in mortalities of any of the study species (Foreman & Alexander, 1998; Grilo et al., 2008). During the data collection some species were more frequently reported as roadkill than actual living individuals that were sighted. It seems clear that roads are affecting mammal abundances to a certain extent. Roads are also vectors for other threats such as the introduction of alien species (unwanted domestic dogs) or diseases and parasites (Foreman & Alexander, 1998; Trombulak & Frissell, 2000). Physical litter, fuel, sound and light pollution are introduced by roads and human developments (Coffin, 2017; Foreman & Alexander, 1998). Roads ease access for humans into natural habitats, thereby exposing more habitat to the negative effects of humans.

Invasive mammals can often gain access to natural habitats through human developments and roads (Trombulak & Frissell, 2000). Both feral dogs and feral pigs that were detected in this study, can drive negative impacts on the ecosystem and other medium-sized mammals. Feral dogs are a major concern for native mammal species. They are well documented predators and deterrents of native fauna globally (Lacerda et al., 2009), but very little is documented in terms of feral dogs in the Fynbos biome. The current study noted feral domestic dogs had been observed impacting 13 medium-sized mammal species and numerous other taxa. It seems most probable that they are a symptom of human presence and show some dependence on humans (Butler et al., 2004). Evidence supporting this was their greater presence on farms further from PA boundaries (Paschoal et al., 2012). Feral dogs were of high concern to many stakeholders and have economic impacts as pests in agriculture, are ecological threats to agro-ecosystems and are a danger to human health and safety (Butler et al., 2004; Lacerda et al., 2009; Paschoal et al., 2012).

Fences are another component of human development that may be constructed on an increasing number of privately-owned properties in the buffer zones in the study area (Landman, 2002). Reasons for their construction might include personal security, protection of crops from pests or livestock or game fencing (Landman, 2002). Most fences have some negative impacts as barriers, and game fences and electrified security fences are complete barriers to many medium to large-sized mammals, restricting movement between farms and the core PAs (Boone & Hobbs, 2004). Natural vegetation may not change in land-cover class but is then inaccessible to some species and could be considered as a form of habitat loss. Fences also enhance human access to linear features via maintenance paths. These are ideal locations to set traps and often wire from fences may be used for the construction of traps/snares, set by opportunistic poachers (Boone & Hobbs, 2004; Lindsey et al., 2012; Nieman et al., 2019). In other incidences, electric fences and fences that are poorly visible have caused mortality of mammals (Hoare 1992). These barriers were a notable feature mentioned by multiple interviewees as to why some mammal species abundances have declined.

The long-term reactions of mammals and specific species to fire disturbances in the Fynbos biome are poorly documented (Rebelo et al., 2019). It is therefore unclear what the exact impacts are to mammals, within areas of increased fire frequencies and greater size areas of habitats burning per year. The many small, individual fires are supposedly better for the survival of most fauna rather than a few, extensive fires. This study found many small fires occurred in most recent periods in the BMC, which could be interpreted positively in terms of mammalian survival. Large areas are however, still burning in total over short periods of time. The probability that refuges typically left available for mammals to flee and take shelter in, are being destroyed is greater than in historic regimes (Driscoll et al., 2010). It can therefore be assumed that changed fire regimes are negatively impacting the survival of medium-size mammals in some way.

Roads are likely concerning barriers in this study, relevant to declining populations of rhebok, genet, weasel and polecat (Klar et al., 2009). This impact requires further study.

Informal poaching with wire snares and other methods is a very direct threat to medium-sized mammals and requires deeper research on the motivations, methods, cumulative effects and solutions. Some farmers' mention that the retaliatory hunting of so-called agricultural damage-causing species needs further investigation into the methods, drivers and what impacts this activity may be having on populations.

4.4 Future recommendations

4.4.1 Priority research and monitoring

As initially stated, the BMC is a key protected area within the Fynbos Biome, but with limited in-depth knowledge available on many of its ecological attributes and the influences of human development on these. This study was able to therefore highlight a variety of aspects to the BMC that require priority research and monitoring to enhance the sustainability of medium-sized mammalian ecology from here-forth. When looking at both the fire regime changes across the landscape for a long historic period and land-cover changes over a shorter period, it is clear that more severe changes occurred in the landscape over the previous 60-year period (until 2017) than the more recent 23-year period until 2013. To gain a better understanding of how land-cover changes may have impacted mammals, one needs more recent shapefiles and more shapefiles from other periods. This would allow one to determine rates at which habitat is being lost and illustrate clearer patterns.

Land-cover studies in the BMC would benefit by considering subclasses within the vegetation class of specific habitat types, vegetation types or structures. Lambert et al. (2006) showed it to be very beneficial to consider mammal abundances, how fragmentation has occurred and what resources remain (Fahrig, 1997). In order to determine the impacts of land-cover change, the habitat requirements of mammals need to be better defined within the Fynbos Biome (Hall et al., 1997).

Specifically looking at the structure of habitat loss and fragmentation in the BMC's landscape may produce more conclusive answers that can be closely linked to mammal abundances. A study on the scale of the entire chain of protected mountain complexes that are adjoined to the BMC, within the Western Cape would aid in better understanding its functional and genetic connectivity to plan for medium-sized mammal conservation (Vogt et al., 2009). Simonsberg MZ's multiple detected differences suggest that a stronger focus be placed on studying the ecology of isolated PA's of similar layouts in the landscape. Future studies should divide the greater study area into portions with similar base-line characteristics as that of the Simonsberg MZ for better comparisons. The effects of low connectivity and accessibility of species from Simonsberg MZ to other habitats needs specific assessments, by determining the permeability of each land-cover type (Eigenbrod et al., 2008). Buffer zones that border more highly human-populated towns need priority research and monitoring as to what negative impacts they are exposed to.

This and many studies have examined fire locally and vegetations fire ecology (Kraaij & van Wilgen, 2014). The reactions of the majority of medium-sized mammals in the Fynbos biome to the local fire ecology is unknown (Parr & Chown, 2003). The next research step will be to examine the biotic effects of fire and how its established regimes are impacting medium-sized mammals. Research needs to find how they can survive in the landscape, in order to determine the correct fire regime to compile management goals (Driscoll et al., 2010; van Wilgen, 2013). When assessing mammal survival during fire events it is important to also consider their use of human land-covers, how these land-covers burn and how they benefit or hinder populations. A study is also needed to examine the social factors of fire in the Cape, with more complete analysis of human-ignitions, like that of Martínez et al. (2007).

Ideally, all study species require an in-depth quantification of abundances and habitat requirements to understand what species can survive in which buffer land-cover types (Brodie et al., 2014). This would allow one to better gage the impacts of land-use and fire regime changes. Species' thresholds for disturbances need to be better understood. Chapter Three indicated grey rhebok and hares are native species that need priority further investigation. Both the scrub and Cape hares need population assessments in the BMC and throughout the Fynbos biome, because of this study's results showing variation of perceived abundances and D'Alton et al.'s (2002) report of scarce sightings in multiple Western Cape regions. Other lagomorph species such as the riverine rabbit *Bunolagus monticularishave* have been flagged as being of conservation concern. It may therefore be beneficial to perform conservation assessments on all hare and rabbit populations in the Western Cape and incorporate in the study Smith's red rock rabbit, (Farfán et al., 2011). As already highlighted by CapeNature, grey rhebok requires continued monitoring of their distributions and population sizes (Birss, 2017).

One of the most important future priorities identified by this study is to focus more resources towards understanding and eradicating feral dogs from natural ecosystem edges. Their sources, drivers and impacts on biodiversity, agriculture, the economy and social impacts on stakeholders are vital to understand to implement efficient control actions. The ethical conflicts behind the eradication of this invasive, domestic species makes research more necessary and important (Lumney et al., 2011).

4.4.2 Management and planning

All management strategies require sound research before implementation. This study suggests in which directions this planning should continue. Legislation that managed to restrict encroachment into both government and privately-owned natural vegetation was well implemented during the study period and must be maintained and continually improved. A suggestion for future farm developments would be to ensure that unique habitats, such as rocky outcrops or renosterveld are not isolated by unnatural land-covers. Any sized, continuous natural habitat into buffer land-covers may extend mammal species ranges. As Pool-Stanvliet et al. (2017) suggested, one should ensure that current habitat is preserved, while trying to convert more land-cover back to natural vegetation and to rehabilitate areas to high capacity habitats (Fahrig et al., 2011). It seems particularly necessary to prevent future clearing of any natural vegetation adjoined to the isolated core zones (like Simonsberg MZ), on privately owned properties. The biosphere reserves, conservancies and other conservation structures in the landscape must remain connected and communications and collaboration between management stakeholders maintained (Mangnall & Crowe, 2003).

The programmes of conservation stewardship initiated by CapeNature and World Wide Fund for Nature (WWF) are well planned initiatives to involve stakeholders in the conservation of their own property and conformity to conservation methods of PAs (von Hase et al., 2010). These should continue to grow, and more farms should be encouraged to join these programmes. Particularly those properties that contain the mentioned rare and functionally important habitat types, like corridors. Furthermore, the inclusion of stakeholders on all social levels in considerations of policy planning, management decisions and conservation actions are often adaptive and efficient strategies for conservation success in buffer zones (Reed, 2008).

In an ideal system one would recommend strategic burning intervals and management according to van Wilgen (2013). However, the BMC's drastic increase in fires seems likely due to accidents and arson from infrastructure development and growing human populations in agriculture, built-up and forestry land-covers and drier vegetation expanses that add to the fuel load (Kraaij et al., 2013; Martínez et al., 2007). Human ignition sources can only be better managed by strong regulations, thorough enforcement of these regulations and education of the processes and risks of fire to all stakeholders. This may not guarantee fire prevention (Clark, 2008; Martínez et al., 2007). Dry vegetation is a result of climate change caused drought and temperature increases, which can only be managed through global sustainable development (Cochrane & Laurance, 2004; Gumbi, 2011; Kraaij et al., 2013). Adaptive management is key for such influential and easily human-influenced disturbances (Driscoll et al., 2010).

Feral dog associations as identified in this study, are closer to humans. They may be controlled through a change of human actions, implementing management strategies, with a focus on prevention. Urgent and strategic attention is required to prevent the spread and further impacts on agriculture, humans and biodiversity (Lessa et al., 2016). Management strategies need to be designed for this species that focus on prevention. After adequate research and monitoring current municipal by-laws need examination and location-specific adaptations. Sterilisation, ownership and containment policies need to be enforced for each type of stakeholder that owns a domestic dog. Stakeholders must be involved in conceptualising management actions and all stakeholders informed about the methods of reporting feral dogs and the correct steps to take when sighted to aid in their population control (Agri SA, 2017; Massara & Chiarello, 2012). It can be communicated to all buffer communities that feral dogs are a newly relevant, widespread problem. The identification of a feral dog must be defined and what simple steps to manage them can be taken by any stakeholder who encounters feral dogs (Mangnall & Crowe, 2003)

Negative impacts from some features of human development detected in this study can be reduced without in-depth research. A way to reduce wildlife-vehicle collisions, is to erect signage to create awareness of what mammals are present in areas for drivers (Kioko et al., 2015). Fences should require ecological impact assessments prior to erection within buffer zones (Boone &

Hobbs, 2004). Education of all stakeholders about the impacts and legalities of hunting with wire snares, dogs and other methods need to be rolled out. Initiation of snare patrols (led by owners or managers) on private properties and protected areas may be necessary in the BMC (Nieman et al., 2019). Overall, awareness and availability of information on what mammals exist amongst humans in these landscapes is needed. It is important that stakeholders understand why mammals are important and how human actions impact them and can have cascading effects on personal influences in the BMC's buffer zones.

4.5 Reference list

Agri SA. (2017). Illegal hunting with dogs – guidelines. www.agrisa.co.za

Barthelmess, E.L. (2014). Spatial distribution of road-kills and factors influencing road mortality for mammals in Northern New York State. *Biodiversity and Conservation*, 23(10), 2491-2514.

Becker, M., McRobb, R., Watson, F., Droge, E., Kanyembo, B., Murdoch, J. et al. (2012). Evaluating wire-snare poaching trends and the impacts of by-catch on elephants and large carnivores. *Biological conservation*, 158, 26-36.

Birss, C. (2017). Mammals. In: Turner, A.A. (Ed.), *Western Cape Province State of Biodiversity 2017* (pp. 191-230). Stellenbosch: CapeNature Scientific Services.

Boone, R.B. & Hobbs, N.T. (2004). Lines around fragments: effects of fencing on large herbivores. *African Journal of Range and Forage Science,* 21(3), 147-158.

Boshoff, A.F., Kerley, G.I.H. & Cowling, R.M. (2002). Estimated spatial requirements of the medium- to large-sized mammals according to broad habitat units in the Cape Floristic Region, South Africa. *African Journal of Range and Forage Science*, 19(1), 29-44.

Brodie, J.F., Giordano, A.J. & Ambu, L. (2014). Differential responses of large mammals to logging and edge effects. *Mammalian Biology*, 80, 7-13.

Butler, J.R.A., du Toit, J.T. & Bingham, J. (2004). Free-ranging domestic dogs (Canis familiaris) as predators and prey in rural Zimbabwe: threats of competition and disease to large wild carnivores. *Biological Conservation*, 115, 369-378.

Caudill, S.A., DeClerck, F.J.A. & Husband, T.P. (2014). Connecting sustainable agriculture and wildlife conservation: Does shade coffee provide habitat for mammals?

Clark, M.F. (2008). Catering for the needs of fauna in fire management: science or just wishful thinking? *Wildlife Research*, 35, 385-394.

Cochrane, M.A. & Laurance, W.F. (2004). Synergisms among fire, land use, and climate change in the Amazon. *Ambio*, 37, 522-527.

Coffin, A.W. (2017). Roadkill to road ecology: A review of ecological effects of roads. *Journal of Transport Geography*, 15, 396-406.

Crooks, K.R. (2002). Relative sensitivities of mammalian carnivores to habitat fragmentation. *Conservation Biology*, 16(2), 488-502.

D'Alton, M.J., Kryger, U. & Suchentrunk, F. (2002). Possible range reduction of Cape hare (*Lepus capensis*) in the Overberg region of the Cape Province. Research Institute of Wildlife Ecology, University of Veterinary Medicine, Vienna, 59-61.

DeFries, R., Hansen, A., Turner, B.L., Reid, R. & Liu, J. 2007). Land use change around protected areas: management to balance human needs and ecological function. *Ecological Applications,* 17(4), 1031-1038.

Eigenbrod, F., Hecnar, S.J. & Fahrig, L. (2008). Accessible habitat: An improved measure of the effects of habitat loss and roads on wildlife populations. *Landscape Ecology*, 23, 159-168.

Fahrig, L. (1997). Relative effects of habitat loss and fragmentation on population extinction. The *Journal of Wildlife Management*, 16(3), 603-610.

Fahrig, L., Baudry, J., Brontons, L., Burel, F.G., Crist, T.O., Fuller, R.J. et al. (2011). Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. *Ecology Letters*, 14, 101-112.

Farfán, M.A., Duarte, J., Vergas, J.M. & Fa, J.E. (2011). Effects of human induced land-use changes on the distribution of the Iberian hare. *Journal of Zoology*, 286, 258-265.

Fehlmann, G., O'Riain, M.J., Kerr-Smith, C. & King, A.J. (2017). Adaptive space use by baboons (Papio ursinus) in response to management interventions in a human-changed landscape. *Animal Conservation*, 20, 101-109.

Foreman, R.T.T. & Alexander, L.E. (1998). Roads and their major ecological effects. Annual *Review of Ecology, Evolution and Systematics*, 29, 207-231.

Golley, F. B., Ryszkowski, L., & Sokur, J. T. (1975). The role of small mammals in temperate forests, grasslands and cultivated fields. In F. B. Golley, K. Petrusewics, & L. Ryszkowski (Eds.), *Small mammals and their productivity and population dynamics* (pp. 223–242). Cambridge: Cambridge University Press.

Grilo, C., Bissonette, J.A. & Santos-Reis, M. (2009). Spatial-temporal patterns in Mediterranean carnivore road casualties: Consequences for mitigation. *Biological Conservation*, 142, 301-313.

Gumbi, D.P. (2011). *The impact of change in climate, human demography, and other social factors on the fire regime of the Kogelberg Nature Reserve*. University of Kwazulu-Natal.

Hall, L.S., Krausman, P.R. & Morrison, M.L. (1997). The Habitat Concept and a Plea for Standard Terminology. *Wildlife Society Bulletin*, 25(1), 173-182.

Hayward, M.W. (2009). Bushmeat hunting in Dwesa and Cwebe Nature Reserves, Eastern Cape, South Africa. *South African Journal of Wildlife Research*, 39(1), 70-84.

Henschel, P., Hunter, L.T.B., Coad, L., Abernethy, K.A. & Mhlenberg, M. (2011). Leopard prey choice in the Congo Basin rainforest suggests exploitative competition with human bushmeat hunters. *Journal of Zoology*, 285(1), 11-20.

Hignett, D.L. (2006). *Feral pigs (Sus scrofa) in the Western Cape: a re-evaluation*. Unpublished Masters' project report, University of Stellenbosch.

Hoare, R.E. (1992). Present and future use of fencing in the management of larger African mammals. *Environmental Conservation*, 19, 160-164.

Kelly, L.T., Nimmo, D.G., Spence-Bailey, L.M., Taylor, R.S., Watson, S.J., Clarke, M.F. et al. (2012). Managing fire mosaics for small mammal conservation: A landscape perspective. *Journal of Applied Ecology*, 49, 412-421.

Kioko, J., Kiffner, C., Jenkins, N. & Collinson, W.J. (2015). Wildlife roadkill patterns on a major highway in northern Tanzania. *African Zoology*, 50(1), 17-22.

Klar, N., Herrmann, M. & Kramer-Schadt, S. (2009). Effects and mitigation of road impacts on individual movement behavior of wildcats. *Journal of Wildlife Management*, 73(5), 631-639.

Kraaij, T. & Novellie, P.A. (2010). Habitat selection by large herbivores in relation to fire at the Bontebok National Park (1974-2009): The effects of management changes. *African Journal of Range and Forage Science*, 27(1), 21-27.

Kraaij, T. & van Wilgen, B.W. (2014). Drivers, ecology, and management of fire in fynbos. In N. Allsopp, J.F. Colville & G.A. Verboom (Eds.), *Fynbos: Ecology, evolution, and conservation of a megadiverse region* (pp. 47-72). Cape Town: Oxford University Press.

Kraaij, T., Baard, J.A., Cowling, R.M., van Wilgen, B.W. & Das, S. (2013). Historical fire regimes in a poorly understood, fire-prone ecosystem: eastern coastal fynbos. *International Journal of Wildland Fire*, 22, 277-287.

Lacerda, A.C.R., Tomas, W.M. & Marinho-Filho (2009). Domestic dogs as an edge effect in the Bras´ılia National Park, Brazil: interactions with native mammals. *Animal Conservation*, 12, 477-487.

Lambert, T.D., Malcolm, J.R. & Zimmerman, B.L. (2006). Amazonian small mammal abundances in relation to habitat structure and resource abundance. *Journal of Mammalogy*, 87(4), 766-776.

Landman, K. (2002). *Gated communities in South Africa: Building bridges or barriers?* International Conference on Private Urban Governance, Mainz, Germany, June 6-9, 2002.

Lessa, I., Guimaraes, T.C.S., Bergallo, H.G., Cuha, A. & Vieira, E.M. (2016). Domestic dogs in protected areas: a threat to Brazilian mammals? *Brazilian Journal of Nature Conservation,* 14, 46-56.

Lindsey, P.A., Masterson, C.L., Beck, A.L. & Romaňach, S. (2012). Ecological, social and financial issues related to fencing as a conservation tool in Africa. In M.J. Somers & M.W. Hayward (*Eds*.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* (pp. 215 -234). Springer Science and Business Media.

Louw, W.J.A. (2010). General history of the South African Forest Industry: 2003-2006. *Southern African Forestry Journal*, 208(1), 79-88.

Lumney, M., Jones, A., Stiles, E. & Waltner-Toews, D. (2011). *Preventative Veterinary Medicine*, 102, 315-320.

Mangnall, M.J. & Crowe, T.M. (2003). The effects of agriculture on farmland bird assemblages on the Agulhas Plain, Western Cape, South Africa. *African Journal of Ecology*, 41, 266-276.

Martínez, J., Vega-Garcia, C. & Chuvieco, E. (2007). Human-caused wildfire risk rating for prevention planning in Spain. *Journal of Environmental Management*, 90, 1241-1252.

Massara, R.L. & Chiarello, A.G. (2012). Is the domestic dog becoming an abundant species in the Atlantic Forest? A case study in southeastern Brazil. *Mammalia*, 76, 67-76.

Matson, P.A., Parton, W.J., Power, A.G. & Swift, M.J. (197). Agricultural intensification and ecosystem properties. *Sciencem* 277, 504-509.

Moran, S., Sofner, S. & Cohen, M. (1987). Control of the rock hyrax, *Procavia capensis*, in fruit orchards by fluoroacetamide baits. *Crop Protection*, 6(4), 265-270.

Nieman, W.A., Leslie, A.J., Wilkinson, A. & Wossler, T.C. (2019). Socioeconomic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa. *Journal of Nature Conservation*, 52, 1-7.

Norton, P.M., Lawson, A.B., Henley, S.R. & Avery G. (1986). Prey of leopards in four mountainous areas of the south-western Cape Province. *South African Journal of Science and Technology*, 16(2), 47-52.

Parr, & Chown, (2003). Burning issues for conservation: A critique of faunal fire research in Southern Africa. *Austral Ecology*, 28, 384-395.

Paschoal, A.M.O., Massara, R.L., Santos, J.L. & Chiarello, A.G. (2012). Is the domestic dog becoming an abundant species in the Atlantic forest? A study case in southeastern Brazil. *Mammalia*, 76, 67-76.

Pool-Stanvliet, R., Dufferl-Canham, A., Pence, G. & Smart, R. (2017). *The Western Cape biodiversity spatial pan handbook 2017.* Stellenbosch: Stellenbosch.

Prugh, L.R., Hodges, K.E., Sinclair, A.R.E. & Brashares, J.S. (2008). Effect of habitat area and isolation on fragmented animal populations. *Proceedings of National Academy of Sciences*, 105(52), 20770-20775.

Radloff, F.G.T. (2008). *The ecology of large herbivores native to the coastal lowlands of Fynbos Biome in the Western Cape, South Africa*. Unpublished doctorate thesis, University of Stellenbosch.

Rebelo, A.J., Rebelo, A.G., Rebelo, A.D. & Bronner, G.N. (2019). Effects of alien pine plantations on small mammal community structure in a southern African biodiversity hotspot. *African Journal of Ecology*, 1-14.

Reed, M.S.C. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation*, 141(10), 2417-2431.

Ricketts, A.M. & Sandercock, B.K. (2016). Patch-burn grazing increases habitat heterogeneity and biodiversity of small mammals in managed rangelands. *Ecosphere*, 7(8).

Rico, A., Kindlmann, P. & Sedláček, F. (2007). Barrier effects of roads on movements of small mammals. *Folia Zoologica*, 56(1), 1-12.

Ruiz, R.F. (2003). *Alternative land uses to forestry in the Western Cape: a case study of La Motte Plantation*. (Unpublished master's thesis). Stellenbosch University.

Shepard, D.B., Kuhns, A.R., Dreslik, M.J. & Phillips, C.A. (2008). Roads as barriers to animal movement in fragmented landscapes. *Animal Conservation*, 11, 288-296.

Sotherton, N.W. (1998). Land use changes and the decline of farmland wildlife: An appraisal of the set-aside approach. *Biological Conservation*, 83(3), 259-268.

Stuart, C. & Stuart, J. (2013). Mammals of Africa. Volume V: Carnivores, Pangolins, Equids and Rhinoceroses. Bloomsbury Publishing, London, United Kingdom.

Stuart, C. & Stuart, M. (2015). Stuart's field guide to mammals of Southern Africa. Struik Nature, Cape Town.

Trombulak, S.C. & Frissell, C.A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology*, 14, 18-30.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I. & Carsten, T. (2005). Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters*, 8, 857-874.

Van Wilgen, B.W. (2013). Fire management in species-rich Cape fynbos shrublands. The *Ecological Society of America*, 11(1), 35-44.

Van Wilgen, B.W., Forsyth, G.G., de Klerk, H., Das, S., Khuluse, S. & Schmitz, P. (2010). Fire-management in Mediterranean-climate shrublands: A case study from the Cape Fynbos, South Africa. *Journal of Applied Ecology*, 47, 631-638.

Vogt, P., Ferrari, J.R., Lookingbill, T.R., Gardener, R.H., Ritters, K.H. & Ostrapowicz, K. (2009). Mapping functional connectivity. *Ecological Indicators*, 9(1), 64-71.

Von Hase, A., Rouget, M. & Cowling R.M. (2010). Evaluating Private Land Conservation in the Cape Lowlands, South Africa. *Conservation Practice and Policy*, 24(5), 1182-1189.

Woodroffe, R. & Ginsberg, J.R. (1998). Edge effects and the extinction of populations inside protected areas. *Science*, 280, 2126-2128.